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Bioremediation of landfill leachate is an attractive alternative to conventional treatment and containment technologies. This study employs a microalgal-bacterial consortium for outdoor, pilot-scale treatment of landfill leachate and utilises the data for techno-economic and sensitivity analysis. Results highlight that capital expenditure and the treatment duration are key parameters to reduce overall treatment cost and make this process economic viable.

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Pathways to economic viability: a pilot scale and techno-economic assessment for algal bioremediation of challenging waste streams

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Abstract:

Waste production and landfilling are a growing problem due to population growth and more affluent societies following a ‘take-make-waste’ linear economy. All landfills generate leachate, which must be detoxified before release to the environment. Current leachate treatment technologies are often energy intensive, relatively expensive and ignore the potential resources which are contained within. The use of adapted microbial consortia for bioremediation of leachate offers not only treatment but an opportunity for reutilisation of lost resources, converting them to fuel, feed and chemical products. In this study, pilot scale experimental data for algal-bacterial leachate treatment in a 300 L photobioreactor is used to perform a techno-economic analysis. The analysis considers the process at larger scales and evaluate where optimisation and future research should be focused to reduce costs and make the treatment financially competitive with existing technologies. Reductions in capital expenditure and treatment time are key areas for cost reductions; potentially saving around 90 % of the total treatment costs.

Keywords:

Leachate, bioremediation, techno-economic assessment, algal-bacterial consortia, pilot scale

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1. Introduction

Higher income and urbanisation seen across the globe means municipal waste production is expected to reach 2.2 billion tonnes annually by 2025 (1). Although the rate of recycling in the UK has been growing over the past decade, 14 Mtonnes of municipal waste was still produced and sent to landfill in 2017 (2). Landfilled waste is contained but not treated or eliminated, which can lead to potential environmental hazards and loss of valuable resources.

Percolation of rainwater through the solid waste and decomposition of components within the waste result in the production of a toxic liquid termed leachate. This liquid effluent is a major environmental concern due to its high chemical oxygen demand (COD), ammoniacal nitrogen (NH_3-N) and heavy metal concentration (3,4). When released into the environment, nutrients within the untreated liquid can cause eutrophication of nearby water sources (5). Furthermore, heavy metals within the effluent, such as arsenic and mercury, can bioaccumulate within the ecosystems, affecting flora, fauna and human health (3,6).

Biological, chemical and physical methods can be used to treat leachate (7), although there is no ‘most appropriate treatment’ available (8). Current treatment methods come with both advantages and disadvantages (summarised in Table 1) meaning they are often used in combination. Problems with current technologies include relatively high expense, energetically demanding, environmentally unsustainable processes and efficiency issues as the characteristics of leachate change (6). For example, air stripping followed by reverse osmosis (8) produces a more concentrated toxic waste product (retentate).

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78 ***Table 1:*** *Advantages and disadvantages of different leachate treatment methods used (8).*

Treatment	Pros	Cons
Combining effluent with domestic sewage	✓ Easy maintenance ✓ Low operational cost	× Low efficiency due to inhibition by organics / HMs
Recycling back through the landfill	✓ Cheap to run ✓ Shortens the stabilisation time of the site	× Inhibition of methanotrophs × High volumes can saturate the landfill causing ponding
Advanced Oxidation Processes (AOP)	✓ Very high efficiency of COD removal ✓ Improves the biodegradability of recalcitrant organic pollutants	× Mainly used in conjunction with other treatments × High energy demand and capital intensity
Air stripping	✓ High NH_3-N removal	× Requires very high pH × Release of gaseous NH_3
Filtration	✓ Eliminates all macromolecules to the filter size	× High expense from filter replacement and pump operation
Reverse osmosis	✓ High recovery rate of various pollutants	× Membrane fouling × Production of an unusable concentrate
Microalgal growth	✓ Can remove a wide range of pollutants at once ✓ Biomass produced can be sold on for future use	× High expense × Low productivity × Requires pre-treatment - Dilution

79

A major issue with the treatment of leachate is the highly variable composition, which depends on the waste type, age and geographical location of the landfill site (9–11). It is very difficult to define a “typical” leachate as even samples from the same landfill site show different characteristics over time (12). A variety of leachates and wastewaters have been characterised within the literature and Table 2 highlights the different key components of each and the variation that can be seen.

The use of photosynthetic microalgae for nutrient and pollutant removal from leachate offers an alternative method of treatment where a useable by-product in the form of biomass is produced. The concept of simultaneous wastewater treatment and algal production was proposed by Oswald *et al.* in 1957 (13,14). Since then an increasing number of studies utilising a diverse collection of microalgal strains have demonstrated that microalgae can remove COD, NH_3-N, orthophosphate and HMs, such as chromium, copper and iron, from these wastewaters with varying degrees of success. As each leachate has its own unique composition and each species or strain react differently to each component there is not one-optimal strain using algal-based treatment of leachate. However, the ability to generate algal biomass through the treatment process is advantageous both from an environmental and economic standpoint. The resulting algal biomass can be converted to a variety of products, including plastics, fuels, fertiliser or animal feed (15–17), improving the economics of the process and producing a closed-loop of nutrient usage where the waste is reutilised rather than disposed of. Integrating bacteria into the process in the form of an algal-bacterial consortium can also aid the treatment process by targeting the bio-degradation of more recalcitrant compounds and hence reducing COD within the leachate (18). Moreover, the algae can capture CO_2 generated by the biological oxidation of organic compounds. The symbiotic relationship has been shown to improve nutrient removal and can make the process more robust to changes in nutrient flux (18).

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There are currently several limitations to algal-bacterial leachate treatment including issues with the dark colour, sub-optimal phosphate levels, toxic organics and very high ammonia levels. The dark colour often associated with leachate affects the photosynthetic potential of algae, adversely impacting the biomass productivity (19). Similarly, the presence of toxic organics and HMs can adversely affect productivity. Algae require both a nitrogen and phosphorous source to grow and leachates can often offer too high concentration of NH_3-N and too little phosphorous. Consequently, leachate is often diluted to 10 % (v/v) to increase transparency and reduce the NH_3-N concentration, together with supplementation with a phosphorous source if necessary (5,19,20).

Another major limitation of algal leachate bioremediation currently is the economic viability of the process. Without any comprehensive techno-economic analyses there is little insight into the economic competitiveness with conventional treatments. Most studies to date state that the use of “wastewater” as a nutrient source will benefit the economics of microalgal production (21–23) but predominantly from the viewpoint of generating lipids for conversion to fuels. Furthermore, where economics are considered, biomass production is the sole focus, not the treatment of wastewater and potential for nutrient recovery.

***Table 2:** Comparison of the different leachate and wastewaters documented within the literature where microalgal-based treatment was investigated. Other than pH, all components are in mg L^{-1} concentrations. *Junk Bay, °Gin Drinkers Bay, LF = landfill leachate, WW = Wastewater, M = municipal, TP = treatment plant, CGW = contaminated ground water, R = raw, RC = recirculated. For [8] the leachates differ by age (a < 5 years old, b 5-10 years old and c is stabilised at more than 10 years old).*

Reference	**Source**	**pH**	**COD**	**Ammonia**	**Nitrate**	**Ortho-P**	**Al**	**As**	**Ba**	**Ca**	**Cd**	**Cr**	**Cu**	**Fe**	**Mn**	**Ni**	**Pb**	**Zn**
Ho *et al.* (9)	LF	5.9 – 6.1	10650											330				
Talalaj *et al.* (12)	LF a	<6.5	>15000	< 400														
	LF b	6.5 – 7.5	4 – 15000															
	LF c	>7.5	< 4000	> 400														
Lin *et al.* (19)	LF	7.6	1280	1345	68.4	5.13												
Eze *et al.* (5)	WW TP	8 - 8.5	280	102.66	1.8	25.6												

Martin & Johnson (24)	LF	405	0.2	2.3				351	2.00			
Kjeldsen *et al.* (25)	LF				0.006	0.08	0.07			0.13	0.07	0.67
	LF				0.005	0.28	0.065			0.17	0.09	0.6
	RC				0.006	0.01	0.04			0.05	0.02	2.2
	LF				0.0002	0.003	0.002			0.028	<0.005	0.2
	LF				0.0004	0.016	0.007			0.084	<0.005	0.36
	LF				0.0003	0	0.034			0.054	0.053	0.085
Cruce & Quinn [19]	LF				0.0036	0.002	0.002			0.062	0.188	5.31
	CGW				0.002 - 0.008	0.033 – 0.085				0.01 - 0.08	0.016 – 0.067	0.003 – 0.011

	LF										0.0002 - 0.018	0.005 – 1.62	0.004 – 0.27			0.0036 - 0.348	0.005 – 0.019	0.05 – 9
	LF										<0.1 - <0.04	<0.01 – 0.05	<0.02 - 0.17			<0.01 – 0.1	<0.04 – 0.13	<0.01 – 0.47
Paskuliakova *et al.* (26)	LF	6.9	145	152		0.5	0.14			111		<0.1	0.4	4.0	1.3	<0.1	<0.001	0.1
	LF	6.3	1505	98		13.8	0.71			445		0.3	<0.1	5.1	0.5	0.16	<0.001	0.3
	LF	7.8	2455	1480		16.5	1.8			295		0.45	<0.1	8.6	0.6	0.19	<0.001	0.2
	LF	8.4	5030	2510		14.8	3.6			413		1.1	<0.1	2.6	0.3	0.5	<0.001	0.2
	LF	7.4	97	122		<0.05	0.2			98		<0.1	>4.0	5.3	1.0	<0.1	0.39	1.7
	LF	7.7	526	506		1.7	0.28			194		<0.1	0.2	3.9	>2	<0.1	<0.001	<0.1
Kang *et al.* (27)	WW		81.45	29.1										0.18	0.118			0.008

Lee *et al.* (28)	M	7.2	295.5	32.5	40.6 ±1.3	7.7 ±0.2												
Ozturk *et al.* (4)	LF	5.6 - 7	35000	2020	2370	5 - 6												
Cheung *et al.* (29)	LF*	7.2	595	724		2.87					<0.01	0.08	<0.01	0.67	0.04	0.07	0.03	0.74
	LF°	7.2	140	147		0.34					<0.01	0.04	0.01	0.88	0.64	0.03	<0.01	0.33
Richards & Mullins	M	8.44 – 8.6	1008				33.56		0.175					15.37	0.27			
Vedrenne *et al.* (31)	LF	8	14680	381		<1		0.233			0.433	<0.1		65	<0.1	<0.1	19.59	0.33

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This study aims to evaluate the economic potential of a microalgal-bacterial consortia for treatment of landfill leachate based on pilot scale experimental data. The experimental data is used as a basis to assess the potential treatment cost and biomass productivity in a 1-ha facility in the UK. A cost-breakdown for both leachate treatment and biomass production are completed for the original data and then five different scenarios. These scenarios are based on potential and realistic changes to either or both financial and operational parameters. Finally, the key parameters (including capital input, operational cost, batch production time and labour costs) are deviated by ± 20 % to explore parameter sensitivity of the overall cost. This analysis is performed to highlight areas of the process for further optimisation and where research and development activities should be focused to make the process economically viable against current commercial techniques.

2. Materials and Methods

2.1. Leachate sampling and algal-bacterial consortia adaption

The leachate utilised for the pilot scale experiment was collected from a leachate pond at the Erin Landfill site in Chesterfield, UK. Samples were taken on the 27th January 2016 and stored in sterile glass containers at 4°C. Analysis of the leachate was conducted and the composition is shown in the supplementary data.

A consortium of algae and bacteria was isolated from the leachate pond by dilution of leachate to 10 % in BBM (Bold Basal Medium) algal media and incubation for 42 days at 25 °C, 150 rpm and 40 μmol $m^{-2} s^{-1}$ light intensity. The consortium was adapted to growth utilising leachate over a period of 24 months through a series of sub-cultures. The ‘adapted consortium’ consists of predominantly *Chlorella vulgaris* microalgae and *Pseudomonas sp.* bacteria.

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2.2. PhycoFlow® pilot scale experiment for leachate treatment

The 300 L PhycoFlow® PBR from VariconAqua (Worcester, UK) was used for the pilot scale treatment of leachate by the adapted algal-bacterial consortia. The PBR is made of Duran borosilicate tubes (5 cm diameter) arranged horizontally in a serpentine formation and connected to a plastic, non-transparent tank. To protect the consortia from extreme temperatures, the PhycoFlow® was encased in a Sunlite Multiwall polycarbonate unit with 83 % light transmission; the setup is shown in Figure 1.

The experiment was conducted between September and November 2017 in Sheffield, UK. The PBR was located outside and utilised natural sunlight as the light source for algal photosynthesis.

Figure 1: *300L Phyco-Flow® set up with and without the polycarbonate casing.*

2.2.1. Inoculum preparation

The PhycoFlow® operation was conducted in batch mode and was initiated with a 20 L inoculum of dense consortia. The inoculum was prepared in BBM containing 10 % (v/v) leachate in a sterile carboy. Initially, 500 mL of consortium was added to 10 L of media and allowed to grow for a week. After this, 2 L of media was added to the carboy each week until

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a final volume of 20 L was achieved. The carboy was continuously aerated at 2 L min^{-1} with ambient air using an ACRO-9630 aquarium pump (Aqualine, UK). The culture was illuminated using artificial lighting (Lumilux cool white fluorescent bulbs) and a regime of 8 hours light:16 hours dark with a light intensity of 70 µmol m^{-2} sec^{-1}. The temperature was maintained at 22 °C throughout the cultivation. The inoculum was grown until an Optical Density at 680nm of 3 was achieved (0.5 g L^{-1} biomass concentration).

2.2.2. PhycoFlow® operation

Before operation, the PBR was chemically sterilised by the addition of 2 % sodium hypochlorite (Alfa Aesar, UK) for 12 hours followed by the addition of 5 % sodium thiosulfate (Fisher Scientific, UK), for neutralisation of the chlorine. The PBR was then drained and filled with fresh, non-sterile tap water. For the experiment, the PBR was filled with 250 L of non-sterile tap water, 30 L of leachate and the 20 L of inoculum, giving a total working volume of 300 L and a working leachate concentration of 10 % (v/v). Additional nutrients in the form of sodium nitrate (0.25 g L^{-1}), dipotassium phosphate (0.075 g L^{-1}) and potassium di-hydrogen phosphate (0.26 g L^{-1}) were added to the PBR to aid the initial growth of the consortia.

During operation, the culture was circulated by a CO4-350/02K 3 phase SS pump (ITT Lowara, UK). The culture was aerated with non-sterile air, daily for four hours using an ACQ-007 air compressor (Boyu, Beijing, China) at a rate of 100 L min^{-1}. The light intensity inside the polycarbonate unit varied between 40 µmol m^{-2} sec^{-1} on cloudy days to 120 µmol m^{-2} sec^{-1} on sunny days. Light intensity was measured using an LI-250A sensor (LI-COR Biosciences, New England, USA). The temperature was controlled inside the unit and maintained between 20 and 25 °C to maintain optimal growth. The temperature was controlled using a 2 kW portable heater (Marko Electrical, UK) and a water-spray cooling system (VariconAqua, UK). The experiment lasted for a total of 42 days, at which point the biomass was harvested using chemical flocculation (chitosan) and microbubble floatation (32).

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2.2.3. Sampling and analysis

The PBR was sampled daily during the initial 7 days of cultivation and then once a week for the remaining 5 weeks. A total of 80 mL was withdrawn for each sample, 5 mL of this was utilised for OD and pH analysis immediately, the remaining was then passed through a 0.22 µm syringe filter (Millex, UK) and then stored at -20 °C until compositional analysis was completed.

The growth of the consortia was followed by OD at 680 nm against a blank of consisting of BBM and 10 % leachate. A standard curve of OD against dry weight was established (data not shown) so that biomass concentration could be determined by the following equation:

$$Dry\ Weight\ (g\ L^{-1}) = 0.1728 \times OD_{680} \qquad R^2 = 0.942 \qquad (1)$$

The pH was measured using a LAQUA B-712 (Horiba, Moulton Park, UK) portable pH probe.

Ammoniacal-Nitrogen concentration was measured in triplicate using the Modified Nessler Method as proposed by Jeong *et al.* (33). Dissolved inorganic phosphate (DIP) was measured using the ascorbic acid method as described by Chian and Dewalle (34). Nitrite and nitrate were measured simultaneously using a colorimetric assay which utilises vanadium (III) for nitrate reduction and detection by the acidic Griess reaction (35).

Removal efficiency (RE, %) and average removal rate (RR, mg L^{-1} d^{-1}) of ammoniacal-nitrogen and DIP were calculated using the following equations:

$$RE = \frac{X_0 - X_t}{X_0} \times 100 \qquad (2)$$

$$RR = \frac{X_0 - X_f}{t_f} \qquad (3)$$

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Where X_0, X_t and X_f are the concentrations at the beginning, time = t and end of the experiment, respectively, and t_f is the total time (in days) of the experiment.

2.3. TEA basic set up

This analysis assumes the production is scaled up to 1 ha, similar to that seen within the literature for small scale algal cultivation (36,37). Each modular PBR unit requires 6 m^2 of floor space. With the allocation of room for a laboratory/office for sampling and inoculum preparation (36) and space between each unit for maintenance access, it is assumed 738 units (230 m^3 culture volume) can be achieved.

As the experimental data comes from the UK during autumn, it is assumed that growth and treatment can be achieved all year round and therefore the facility is operational 360 days of the year. To maintain coherence with the experimental data the same operational procedure of batch culture was chosen. With the 42-day growth time used in the experiment this equates to a full 8 batches being produced annually, with additional time being used for cleaning and maintenance of the facility.

The biomass productivity and nutrient / HM removal results from the experiment are used to determine the outputs from the system in the form of: a) wet biomass which can be sold on for further downstream modification and b) the remediation of leachate, allowing water to be discharged. No downstream processes are included within the scope of this assessment, as shown in Figure 2.

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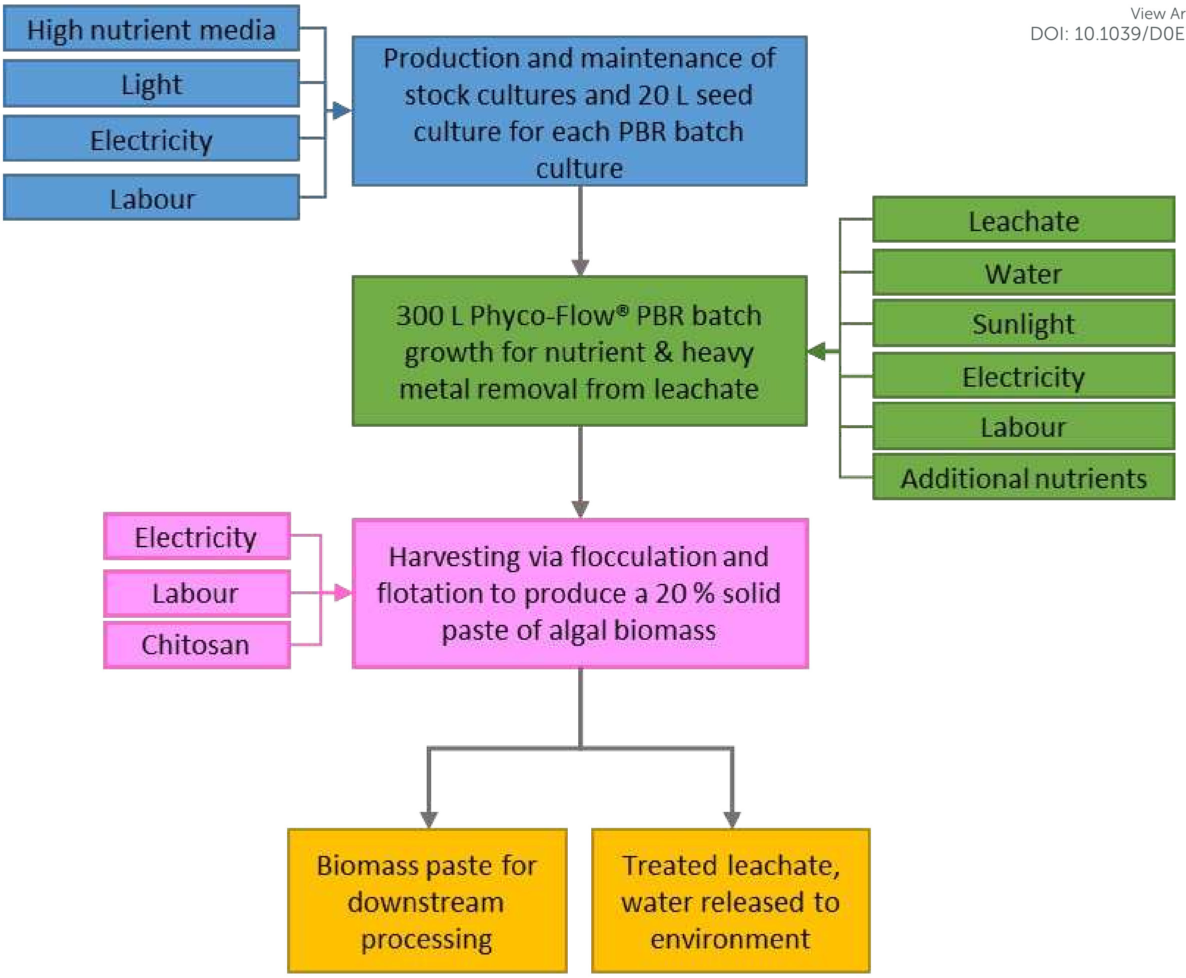

Figure 2: *Flow diagram of the processes and inputs used in the economic analysis. All stages from inoculum preparation to the production of a thick algal paste are included. To highlight the cost of leachate treatment rather than production of algal products downstream processes such as hydrothermal liquefaction, heavy metal removal, carotenoid extraction and drying have purposefully been removed from the system boundaries, assuming the biomass will be sold on for further processing.*

2.4. Financial assumptions and considerations

For a theoretical facility such as this one, a number of key assumptions must be made in regards to the finances and set up (38–41). Parameters such as contingency, maintenance budget, depreciation schedule, discount rate and construction period were all taken as the average values used within the literature, as shown in Table 3. The cost of industrial land (£

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ha^{-1}) and electricity (£ kWh^{-1}) were calculated as the average values for these in the UK (42,43). The price of water and sewerage were taken as the standard rates of Yorkshire, UK (44). Salaries for labour expenses were calculated based on the UK average for each job role and the overheads as 60 % based on the literature (45–47). Financing and tax deductions are not included in this assessment (48).

Table 3: *Financial assumptions made for the TEA.*

Item	Value	Reference
Project lifetime	20 years	(46,47,49–54)
Depreciation of assets	Straight line, no salvage value	(36,49,55)
Maintenance budget	5 % of direct capital cost	(36,46,47,49,56,57)
Contingency allowance	15 % of direct capital cost	(58,59)
Labour overheads	60 % of labour cost	(45–47)
Inflation	2.7 %	UK average 2014-2017 (60)

2.5. Capital Expenditure (CapEx)

The CapEx constitutes direct and indirect costs as well as the residual value of depreciable assets owned:

$$CapEx = TDC + TIC - DepValue \tag{4}$$

Where TDC and TIC are total direct and indirect costs, respectively and DepValue is the current value of depreciable assets owned.

The direct costs include land, buildings, and major equipment cost (MEC):

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$$Total\ Direct\ CapEx\ (TDC) = Land + Buildings + MEC \tag{5}$$

For costs relating to MEC, the prices obtained from industry for the products used in the experimental work are scaled up to the facility size. Due to the modular design of these PBRs, economies-of-scale are not considered for the baseline scaled-up application (46,55,61,62), and therefore the costs for more PBRs can be simply calculated if required. The cost of land, as previously mentioned, is based on UK average prices for industrial land and the building costs are taken from Tredici *et al.* (2016) and converted from Euros to Sterling and then scaled to 2019 equivalent values (36).

The indirect component of CapEx includes monies set aside for contingency planning, assumed to be 15 % of direct costs here to match the novelty of this process, and a maintenance budget set at 5 % of the direct costs:

$$Total\ Indirect\ CapEx\ (TIC) = Contingency + Maintenance \tag{6}$$

Depreciation of assets and any residual value in those with longer lifespans than the project time are considered within the CapEx calculations. All physical property was given a lifespan based on manufacturer information and it is assumed the value of each item depreciates linearly over this time. It was also assumed that the salvage value, the value of the item at the end of its lifespan, was zero. The only exception to depreciation was the land purchased for the facility. The land value is assumed not to depreciate or appreciate over the course of the project lifetime.

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DOI: 10.1039/D0EW00700E

2.6. Operational Expenditure (OpEx)

The total annual OpEx is calculated to include three major items: direct cultivation OpEx (DCO), annual labour OpEx and indirect OpEx:

$$Total\ OpEx = DCO + Labour + Indirect\ OpEx \tag{7}$$

The OpEx is only applicable to operational years of the facility meaning that, assuming a 1.5-year construction and installation period, for 18.5-years of the total project lifetime there is 100% facility operation. In the half year of operation, the DCO and indirect OpEx are scaled accordingly, however labour costs are not and are assumed to be of full value due to the requirement of staff for installation, commissioning, and training prior to production.

The DCO is calculated using the experimental operational data for the single PBR and then scaled accordingly (as seen for CapEx). This element includes all the nutritional, water, heat, and energy inputs required for the cultivation and harvesting of the biomass.

As the facility contains 738 PBR units, labour costs are included in this TEA. Within the literature there is no consistent method for labour cost inclusion and the assumptions made differ dramatically between publications (45–47,63–66). Here, the UK average salary for scientific technicians (£22,000 per annum (67)) and laboratory supervisors (£30,000 per annum (68)) are used with a 60 % overhead for additional services (45,46). The TEA takes four technical staff and one supervisor into account when calculating labour costs. The base cost for labour is £188,000 per annum although this increases with inflation over the project lifetime.

The indirect OpEx is included to cover the cost of maintenance and insurance which will be charged annually. Both the consumables and insurance costs are assumed to be percentages (10 % each) of the combined total of DCO for all units and annual labour OpEx:

$$Indirect\ OpEx = 20\% \times (DCO + Labour) \tag{8}$$

2.7. Outputs

There are assumed to be two main outputs for this process: 1) wet biomass paste (which can be sold on for further downstream processing) and 2) the treatment of the leachate and release of clean water to the environment.

An overall project wide cost-breakdown was created to highlight how cost intensive each process is. Alongside this, a minimum selling price (MSP), the lowest price at which the biomass paste can be sold and the project break even (£ kg^{-1}), was determined using Eq. 9:

$$MSP\ (£\ kg^{-1}) = \frac{CapEx + OpeX - Revenue}{Biomass\ Yield} \tag{9}$$

A treatment cost (TC) for the leachate treatment was also calculated, based on the same principle as the MSP but for a m^3 volume of leachate:

$$TC\ (£\ m^{-3}) = \frac{CapEx + OpeX - Revenue}{Volume\ of\ leachate\ treated} \tag{10}$$

Furthermore, the operational cost and capital investment requirements for each batch cultivation was analysed.

2.8. Scenarios

Once the baseline results were obtained, five different financial and operational scenarios were tested to ascertain if there was a positive change in MSP, TC or batch cost of treatment, and which parameters were most affected. This method of scenario-based analysis is seen throughout the literature for both algal bioremediation and biofuel production (40,46,73–

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76,47,49,55,61,69–72). It highlights how different financial, political, and technological situations can drastically effect whether an algal-based project is economically feasible. The scenarios chosen in this work posit reasonable improvements in algal productivity, operational management and/or investment requirements, all of which are reasonable near-term goals due to the pilot scale nature of the work.

In the first scenario the capital expense of the facility is reduced. The price of MEC input into this assessment are accurate for the purchase of one PBR unit and fitting, not for a hectare facility full of them. Cost reduction to MEC components is done through economies of scale with the average exponent factor of 0.6 (77). Communication with industrial partners clarifies that this reduction is within reason. In Scenario 2 the operational expenditure is considered, particularly the cost of electricity. The leachate used in this experiment was stored at 4 °C prior to use but in reality leachate can have temperatures of around 35 °C (78); therefore in this scenario it is assumed that the heat from the leachate can be utilised for heating the culture, rather than using an external source and this is omitted. The additional nitrogen added to the PBR has also been removed as further lab scale experiments have shown the consortia's ability to grow without this additional input (data not shown). Scenario 3 combines the impact from both the first and second scenarios, where costs can be cut in both capital and operational settings at once. Scenario 4 looks at how improving the efficiency of the biological processes can affect the cost. In this scenario, the batch time required is halved, theoretically based on further strain and consortia development in the lab. This doubles the annual production for the facility. While the operational costs (except labour) will inevitably increase in this case the capital investment should not.

Scenario 5, the final scenario combines the changes made in both Scenarios 3 and 4, an overall 'best-case' scenario. All five scenarios were input to the TEA model and the cost breakdown, MSP TC and NPV (where applicable) were calculated and compared.

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2.9. Sensitivity analysis

Further to the scenario testing, key parameters highlighted from the baseline analysis were analysed individually for their effect on the overall cost of leachate treatment. The MEC, operation cost, labour cost and batch production times were varied by ±20 % from their baseline values to show how they impact on the overall cost.

3. Results and Discussion

The TEA presented in this work is based on experimental, operational, and cost data obtained for the 300 L PhycoFlow® PBR unit. The aim of the process is the simultaneous treatment of landfill leachate and production of algal biomass for further downstream processing and utilisation.

3.1. PhycoFlow® experimental results

A pilot scale, batch experiment for the treatment of landfill leachate and simultaneous algal biomass production was conducted using the 300 L PhycoFlow® PBR. The growth rate and reduction of key nutrients was followed throughout the experiment. The key results are presented in Table 4 and Figure 3.

Table 4: *Experimental results used in this TEA*

Parameter	Value	Units
Time of the batch run	42	Days
Culture productivity (average)	0.124	g L^{-1} day^{-1}
Final biomass concentration	2.4	g L^{-1}
Wet biomass harvested	0.7215	kg PBR^{-1}
Harvesting efficiency	95%	% removal
Moisture content of harvested biomass	80%	%
Leachate ammonia removal efficiency	86%	%
Leachate DIP removal efficiency	100%	%

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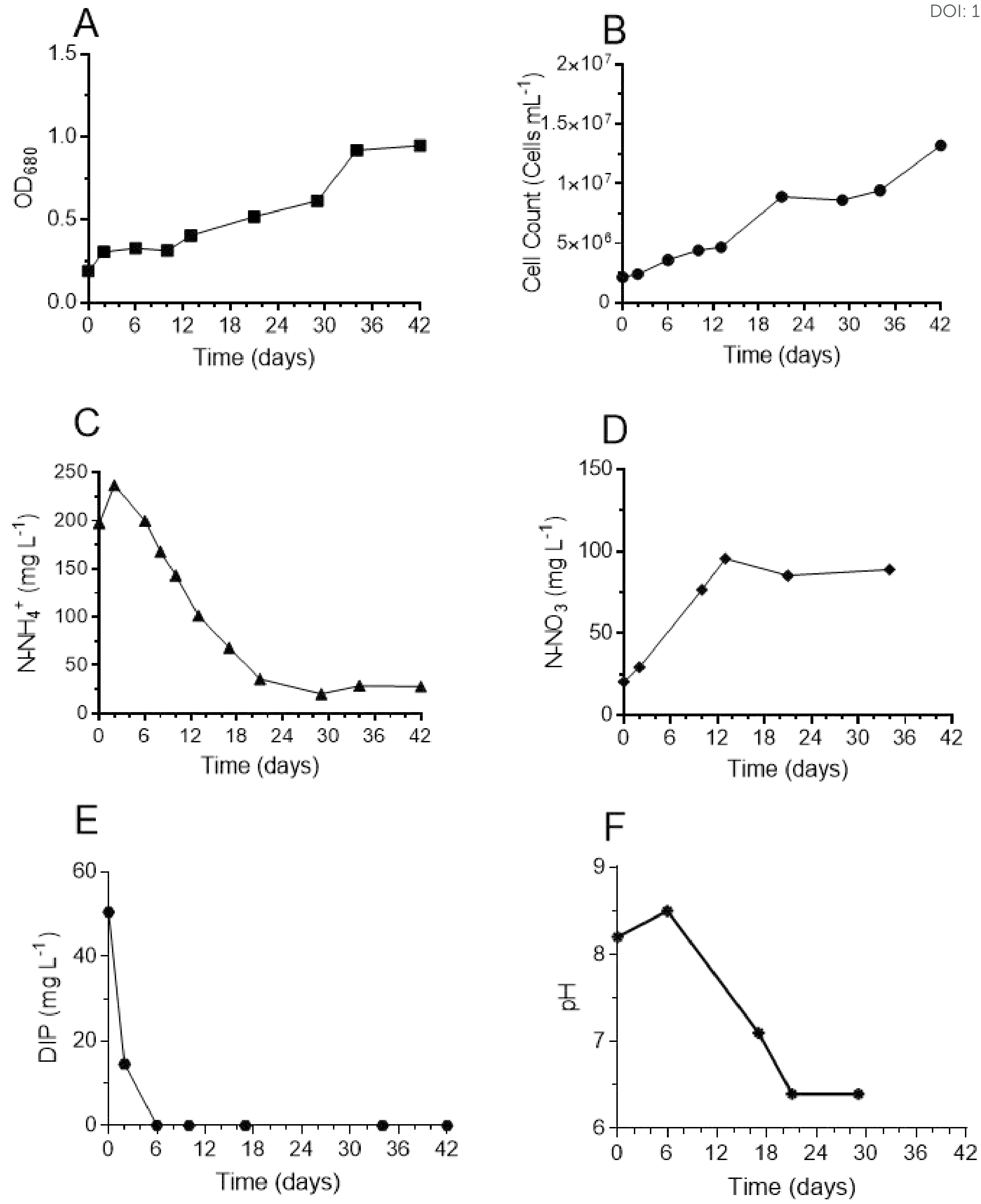

***Figure 3:** PhycoFlow PBR batch experiment results*

3.1.1. Growth of algae and bacteria and pH variation

The microalgal cell numbers increased steadily over the course of the experiment, although there were fluctuations in the rate of growth, particularly noticeable through an acceleration between days 13 and 21 and between days 34 and 42 (Figure 3.B). There were periods where OD_{680} increased, but not at the same time points as microalgal cells (Figure 3.A and 3.B). This is not entirely unexpected as peaks in heterotrophic bacterial activity have previously been shown to follow peaks in primary production in algal-bacterial cultures (79), a phenomenon potentially occurring within the PBR in three different stages. The temperature fluctuated in the reactor from 19 °C to 26 °C. The warmest period was between days 13 and 21 which coincides with an increase in growth rates (based on cell counts and OD_{680}).

Although phototrophic microalgal growth and/or excretion of basic metabolites from biodegradation of organic matter often increases the pH of the media (80), there was an overall decrease in pH from 8.2 on Day 0 to 6.4 in the PBR over the course of the experiment (Figure 3.F). The complexity of the leachate composition as well as the microbial consortium within the PBR means there are many factors which could impact on pH changes, including microbial activity generating CO_2 and volatile fatty acids.

3.1.2. Nitrogen removal

The ammoniacal-nitrogen (NH_3-N) concentration at Day 0 was 197 mg L^{-1}. There was an initial increase in concentration to 237 mg L^{-1} in the first two days, before the concentration reduced steadily to below 20 mg L^{-1} at day 29 (Figure 3.C). The initial increase was likely due to bacterial ammonification of other nitrogen sources within the complex leachate. This is evidenced by an OD_{680} increase in this period, which was not followed by microalgal cell count, implying bacterial growth.

The dissociation constant, pKa, of the ammonia/ammonium reaction is approximately 9, depending on a reaction conditions (temperature, salinity etc.). This pKa value and the low pH of the diluted leachate (<8) mean that ammonium ions (NH_4^+) were dominant over ammonia (NH_3) within the PBR. Ammonium ions have lower toxicity and volatilisation rates in comparison with ammonia, allowing for greater overall removal by the microalgae. 86 % of the ammoniacal-nitrogen was removed from the PBR over the 42 day cultivation period, with a relatively high average and maximum removal rate of 7.7 and 14.0 mg L^{-1} day^{-1}, respectively. Current understanding of using microalgae for ammonia removal from leachate varies depending on the species, cultivation vessel design, aeration, mixing as well as the pH, temperature, and photoperiod (81–83). A previous study by Martins *et al.* reported 75 – 99 % removal of the ammonia from landfill leachate using stabilization ponds (83). Interestingly, a nitrogen balance revealed that under the conditions of the continuous treatment system tested, 64 – 79 % was contained within dead or inert settled algal cells, whereas 1 – 6 % was assimilated into live algae (*Chlamydomonas* genera), with 12 – 27 % of removal by volatilization. The ammonia volatilization rate was not measured during this experiment, however considering the design of the PBR (closed system), pH, reduced flow rate, short aeration period (4 hours per day) and temperature range (19 – 26 °C) during the experiment, the volatilization rate is expected to be lower than in an open pond (83). There was evidence that bacterial based nitrification had taken place during the first 10 days of cultivation as levels of nitrate and nitrite increased (Figure 3.D).

3.1.3. Dissolved inorganic phosphate removal

Concentrations of bioavailable phosphate in landfill leachates are generally quite low, and as expected, the DIP concentration in the 10 % dilution of leachate was only 0.061 mg L^{-1}. Like other landfill leachate treatment studies using microalgae (84), P-supplementation was undertaken to avoid P-limitation. After supplementation, 50.5 mg L^{-1} DIP was measured on day 0 of the experiment. Within 6 days, DIP was almost below detection limits in the leachate (Figure 3.E), with an average removal rate of 8.4 mg L^{-1} d^{-1}.

Although the majority was assumed to be consumed by microalgae, most of the growth took place after 12 days: implying the use of luxury-P or alternative sources of P after this time. It is known that phosphate can precipitate in microalgal cultures where the pH is higher than 8 and it should be considered that some may have precipitated during the first few days of

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cultivation when the pH was recorded above 8. Our results do indicate that although algal growth is possible when DIP levels are close to zero, additional provision of this essential element would likely increase biomass accumulation within the PBR.

3.2. Baseline Economic Scenario

The baseline scenario was conducted using the experimental results as the input. The cost breakdown of the process is shown in Figure 4. The MEC contributes the largest cost at 49 % of the overall cost. Within this (Figure 4B), the PBR unit contributes the largest proportion at 91 % of the MEC. This highlights that the use of a modular, glass system may not be ideal for low value applications such as leachate remediation. To avoid the high-capital investment associated with such PBRs, alternatives made of plastic could be utilised (36). These units however have a much shorter lifespan to the glass counterparts which may reduce the cost-reduction potential. Of course, this baseline assessment does not consider the potential for wholesale bulk trading discounts to the capital price of the PBR due to a large order of units, this is therefore addressed in future scenarios.

The second largest cost was the DCO, contributing 23 %, followed by indirect CapEx at 13 %. Within the DCO value, electricity demand is the largest contributing factor. The requirement for heating the units over autumn and winter months, where average temperatures in Sheffield is 5.5 °C (85), is a major factor here (Figure 4C). The sourcing of process heat from elsewhere would be highly advisable to reduce these costs. The electricity requirements for the pump operations also contributes significantly to the DCO, which is typical of PBRs with similar designs (86).

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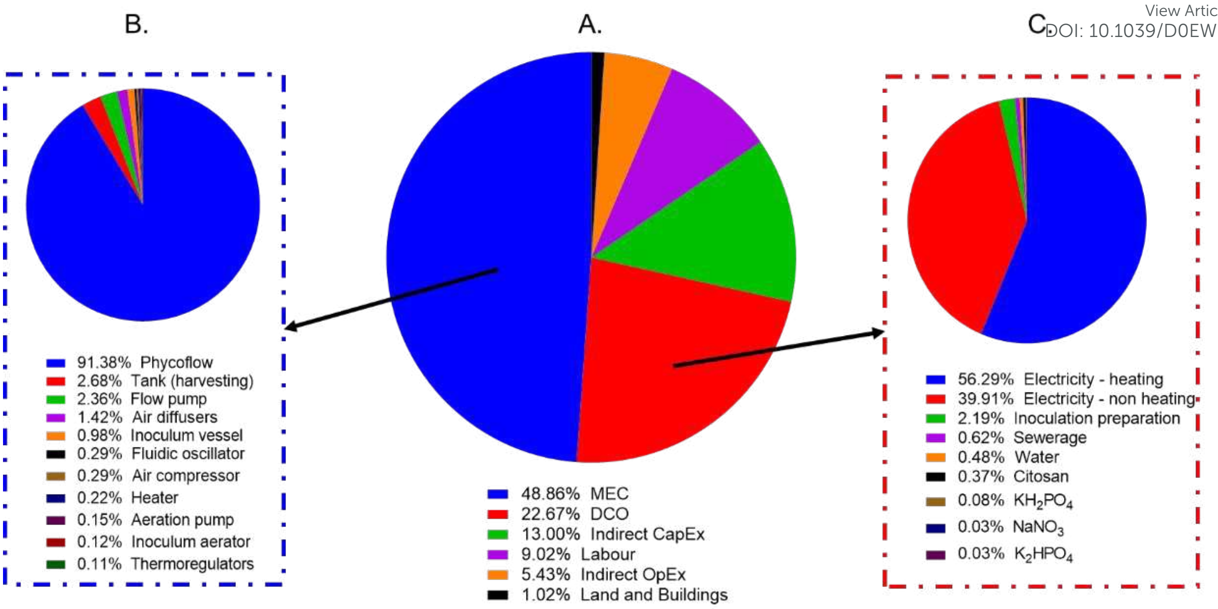

Figure 4: *Cost breakdown for the algal leachate treatment facility. A. Overall cost-breakdown of the facility. B. Further cost breakdown of the MEC component. C. Further breakdown of the DCO component.*

In this base case scenario, it is clear that optimisation of the energy demand and capital input are key for economic viability.

3.3. Scenario results

The five scenarios were input to the model used for the baseline estimations. A cost breakdown along with values for overall cost, capital expense and operational expense were produced for each scenario. The cost breakdown and change in MSP from the baseline for each scenario is shown in Figure 5.

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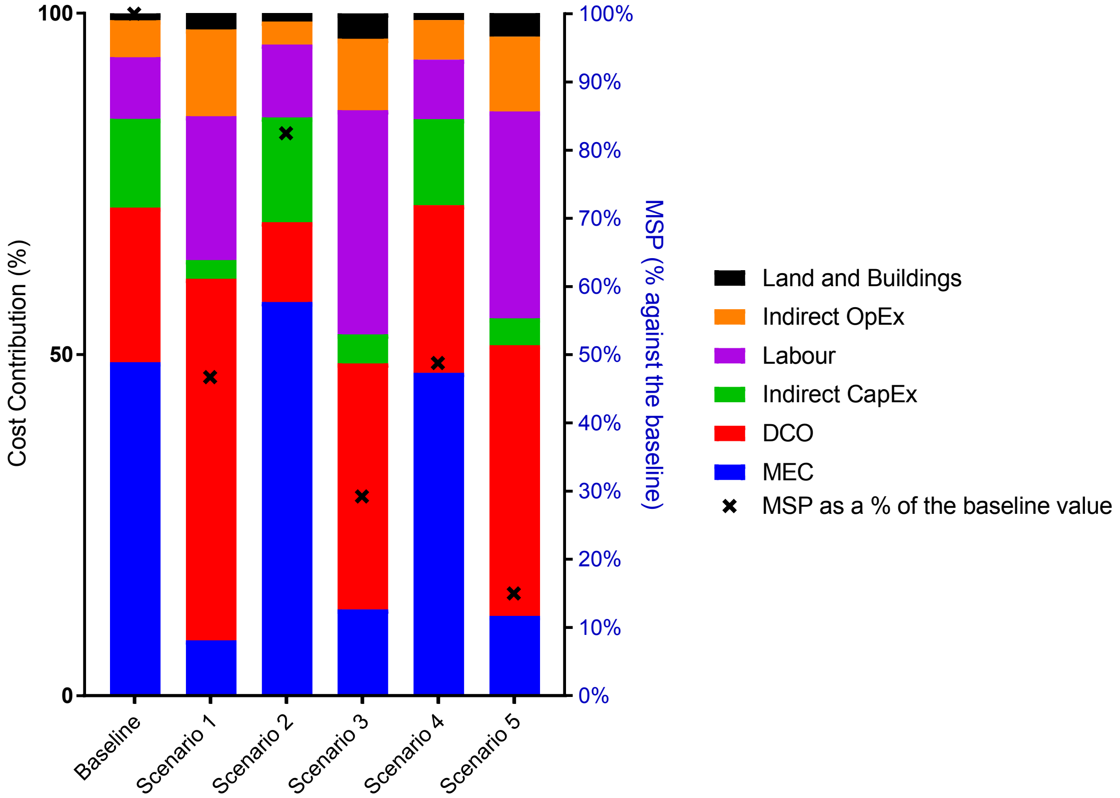

***Figure 5:** Cost breakdown (Bars and Left y-axis) and percentage change of MSP against the baseline (X and Right y-axis) for each of the 5 financial and operational scenarios.*

In Scenario 1, the cost of major equipment such as the PBR and aeration equipment were reduced through economies of scale, based on communications with the manufacturer. This change reduced the overall costs so that the MSP dropped by approximately 53 %. The MEC contribution to the cost was reduced from 49 % to 8 % and as a result the DCO's contribution to cost rose from 23 % to 53 %, becoming the largest contributing factor.

Scenario 2, where the cost of operational parameters such as nutrient input and electricity demand were reduced, shows MSP was reduced by approximately 18 %. The contribution to costs of the DCO was reduced by ~ 50 % from 23 % contribution to 11 %, also causing the indirect OpEx to be reduced and the capital proportion (MEC etc.) to increase.

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In Scenario 3 both the effects of reducing capital and operational costs were assessed together. This reduced the overall MSP by 70 %, with DCO being the primary contributor to the overall cost at 36% with labour expenses as the second largest cost at 33 %.

In Scenario 4, improvements in the consortia treatment and growth were considered, assuming the same biomass concentration and treatment quality can be achieved in half the time currently used in the base model. This resulted in the largest reduction to the MSP value thus far with a 51 % decrease from the baseline values. Due to the increased capacity seen in this scenario and therefore the larger requirement for reagents, the DCO increased by 9 %, while the MEC remained the largest contributing factor overall.

In the final, best-case, Scenario 5, the contribution of all reductions / operational adjustments resulted in the MSP dropping to 15 % of the original baseline value. This was achieved with basic operational and capital modifications. Further research and development into both the experimental methodology and expenditure, both in capital and operational sense, could further reduce these values causing the process to become economically feasible.

To further understand the costs associated with algal leachate treatment, the cost of each scenario is broken down in to capital and operational expenditure and these values are shown in Table 5. As mentioned previously, the main proportion of costs (for all scenarios) is attributed to either the capital investment required or the DCO. When the capital investment is removed (as seen in the literature (22,71,87)) and the operational costs are presented, the cost of each batch culture is £170 for the baseline and £60 in the best case scenario.

Table 5: *Split CapEx and OpEx results for the cost of producing biomass (COPB), a single batch operation and leachate treatment for each scenario*

	Baseline	**Scenario 1**	**Scenario 2**	**Scenario 3**	**Scenario 4**	**Scenario 5**
CapEx (% of cost)	63 %	13 %	74 %	21 %	61 %	19 %
OpEx (% of cost)	37 %	87 %	26 %	79 %	39 %	81 %
Total COPB (£ kg^{-1})	530	250	440	160	260	80
CapEx contribution to COPB (£ kg^{-1})	310	20	310	20	140	10
OpEx contribution to COPB (£ kg^{-1})	220	230	130	140	120	70
Total Cost per Batch Operation (£ batch^{-1})	410	190	340	120	210	70
CapEx per Batch Operation (£ batch^{-1})	240	20	240	20	110	10
OpEx per Batch Operation (£ batch^{-1})	170	170	100	100	100	60
Total Cost of Leachate treatment (£ m^{-3})	12,280	5,740	10,140	3,590	6,000	1,840
CapEx contribution to Leachate treatment (£ m^{-3})	7,090	540	7,090	540	3,340	260
OpEx contribution to Leachate treatment (£ m^{-3})	5,190	5,200	3,050	3,051	2,660	1,580

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3.4. Single parameter analysis

Further to the scenario analysis, key parameters were taken individually and altered by ± 20 % of their original value to highlight how sensitive the MSP is to each parameter. The results in Figure 6 show that the number of batch cultures/treatments which can be achieved annually has the most profound effect on the overall treatment price. Reducing the residence time required for nutrient removal and biomass growth will increase the number of batches each PBR can produce annually, ultimately reducing the cost of the leachate treatment significantly. In this example, the residence time has been reduced from 42 days to 33, allowing 10 batches to be completed annually by each unit rather than 8 (20% increase in the number of batches performed). This small increase in productivity allows the MSP for the biomass produced to be reduced to 80% of the cost in the original assessment. This suggests that improvements in treatment efficiency and/or changing to a semi-continuous method may be advantageous when trying to optimise against costs.

The MEC cost was the next parameter to cause the most significant change from the baseline. Reducing MEC by 20 % allowed the MSP to be reduced by 10 % to 90 % of the original baseline value. As previously mentioned, the usage of a lower capital-intensive reactor would help lower these costs further. The DCO and labour both affect the treatment cost in a similar manner to one another, with little change either side of the original value, ± 4.7 and 3.4 %, respectively.

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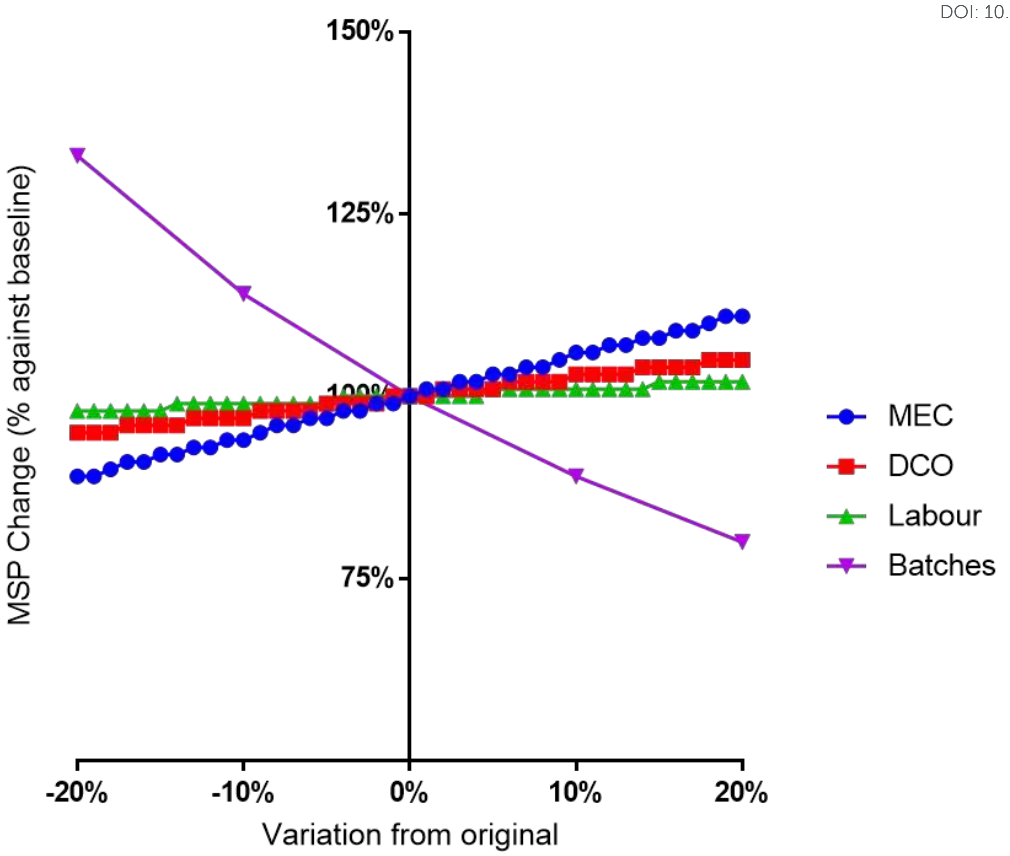

Figure 6: *Changes in MSP from the original baseline value when key parameters were altered by up to ± 20 % either side of the original value.*

4. Conclusions

Pilot scale experimental data for algal-bacterial leachate treatment was used to perform a TEA of the bioremediation process. The initial results show that operational costs for each batch culture/treatment is approximately £170 when no optimisation or cost reduction strategies are put in place. This would need to be reduced to improve economic viability of the process. However, the use of microalgae for leachate remediation can be advantageous if the resulting biomass can be utilised or metals recovered. Useful product(s) can be derived from algal biomass and intensive research is currently being undertaken to broaden this to different markets including: food, plastic alternatives, fertilisers, fish and aquaculture feed and biofuel (both biodiesel or direct burning), reutilising components which are otherwise lost in landfill.

Other treatment methods, such as reverse osmosis, do not currently offer this advantage and still lead to the production of a toxic retentate waste.

The scenario-based analysis highlighted that reductions in both CapEx and OpEx are key to make algal-bacterial leachate remediation feasible. Applying economies of scale to PBR purchases in line with manufacturer quotations and reducing the reliance on fresh water and bulk chemicals for supplementation can reduce the overall cost by 85 % against the baseline.

The sensitivity analysis highlighted that increasing the number of batch treatments that can be achieved annually by either increasing algal-bacterial growth rates or moving to a continuous treatment method can reduce the retention times required and would yield the greatest reduction in overall costs. While this analysis is theoretical in nature it provides key insight to where research should be focused to achieve a more financially feasible algal bioremediation technology.

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Conflicts of interest:

There are no conflicts to declare.

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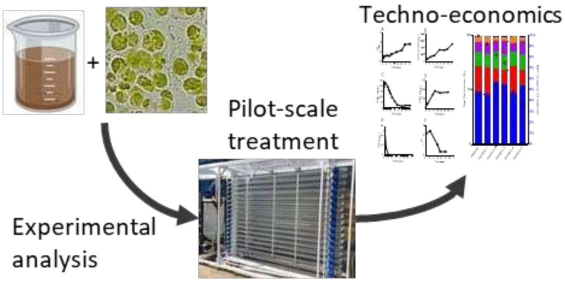

A microalgal-bacterial consortium was used for pilot scale bioremediation of landfill leachate. A techno-economic analysis was conducted using experimental results to provide a pathway for economic viability.