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Review

Evaluating Optimized Potentially Net-Negative Algal Systems and Biochar Production in Global Carbon Sequestration Strategies

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Abstract: To meet the 1.5 °C climate target established by the Paris Agreement, the deployment of Negative Emission Technologies (NETs) is imperative. Algal systems, comprising microalgae and macroalgae (seaweeds), offer one of the most efficient biological pathways for Carbon Capture and Utilization (CCU). However, the long-term sequestration potential of algal biomass is often limited by rapid decomposition. The integration of algal cultivation with biochar production via thermochemical conversion (pyrolysis) offers a synergistic approach to "net-negative" carbon accounting. This review evaluates recent advancements in optimized algal cultivation, focusing on metabolic engineering and photobioreactor design to maximize CO_2 uptake. Furthermore, it analyzes the conversion of algal residues into biochar, assessing its stability, nutrient-recycling potential, and soil-sequestration capacity. Key constraints persist: drying wet algal biomass is energy-intensive, wastewater-grown biomass can carry heavy metals into biochar, and most integrated systems have yet to move beyond the laboratory scale. Most reported productivities, biochar yields and stability estimates come from laboratory experiments or models, and pilot-scale evidence for complete algae-to-biochar chains is still scarce. Where biochar storage is included, life cycle assessments range from net removal (e.g., −3.71 to −2.17 kg CO_2-eq per kg of product for a seaweed biorefinery with biochar) to net emissions, depending on energy sourcing, drying demand and system boundaries, so the net-negative status of these systems is conditional rather than inherent. By synthesizing Life Cycle Assessment (LCA) data and Techno-Economic Analysis (TEA), this review highlights the challenges and opportunities in scaling these integrated systems as a potential contributor to global carbon management strategies.

Keywords: algae; biochar; carbon sequestration; net-negative emissions; pyrolysis; sustainable development

1. Introduction

Anthropogenic greenhouse gas (GHG) emissions continue to increase atmospheric carbon dioxide concentrations and remain the dominant driver of global climate change [1,2]. Recent assessments indicate that energy-related CO_2 emissions reached approximately 37.8 Gt CO_2 in 2024, while atmospheric CO_2 concentrations exceeded 422 ppm, nearly 50% above preindustrial levels [3,4]. Continued emissions at this scale threaten the ability to limit global warming to the 1.5 °C target established under the Paris Agreement, increasing the likelihood of severe ecological, economic, and societal impacts [5,6]. While rapid emission reductions remain the most powerful tool to reach long-term climate goals, mitigation alone is increasingly recognized as insufficient [5,7]. Consequently, substantial attention has shifted toward Negative Emission Technologies (NETs) and Carbon Dioxide Removal (CDR) strategies that can remove previously emitted CO_2 from the atmosphere [8,9]. The IPCC Sixth Assessment Report highlights carbon removal as an important component of pathways limiting warming to 1.5 °C, with recent estimates suggesting that approximately 7 to 9 Gt CO_2 year^{-1} of atmospheric removal may be required by mid-century to offset residual emissions from sectors that are difficult to decarbonize [7,10]. This figure refers to all removal methods combined; the share that any single pathway could supply is limited by the land, water, nutrients and energy it requires (Section 6).

Current CDR approaches include afforestation and reforestation, bioenergy with carbon capture and storage (BECCS), direct air capture (DAC), enhanced weathering, and soil carbon sequestration [11,12]. These methods differ widely in cost, technological maturity, storage duration, and resource requirements. Afforestation and reforestation are the most mature (TRL 8–9) and least expensive options, at approximately $0–240 per tonne CO_2,

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but they compete with food production for land and water resources, and the stored carbon can be lost to wildfires, pests, or land-use change [7,12]. Engineered systems such as BECCS and DAC are more recent technologies (TRL 5–6) [12]. When paired with geological storage, these technologies can hold CO_2 for 10,000 years or more [10]. However, BECCS requires land to grow biomass, and its costs range widely, from roughly $15 to $400 per tonne CO_2. DAC avoids land demand but requires large amounts of energy, which keeps costs at around $100–300 per tonne CO_2 [12]. Enhanced weathering offers similarly durable storage in minerals but is at an earlier stage of development (TRL 3–4) and carries impacts from mining and transport [11,12]. Terrestrial biochar, a soil carbon sequestration strategy, is relatively inexpensive at $10–345 per tonne CO_2 and is capable of storing carbon for centuries, yet its potential is often limited by how much sustainable biomass is available [12,13].

Algal biomass offers one way around this constraint. Rather than relying on terrestrial feedstocks like crop residues and wood, biochar can be produced from algae. The costs and carbon removal potential of algal systems are not yet well established because they remain energy-intensive and few systems have been demonstrated beyond the laboratory scale. Nevertheless, algal systems have attracted growing interest because of their high photosynthetic efficiencies, rapid growth rates, and ability to utilize non-arable land and non-potable water resources [13]. Unlike terrestrial vegetation, which is constrained by soil availability and freshwater demand, algae can be cultivated in saline water, brackish environments, municipal wastewater, and other marginal resources, reducing competition with agricultural systems for land and water [13,14]. Feedstocks can therefore come from a range of production systems, giving algae-to-biochar systems the flexibility to match local resources. For example, ocean-based strategies like large-scale seaweed cultivation offer large removal potential with relatively low input [15]. Microalgae also demonstrate high removal potential with photosynthetic efficiencies that exceed those of most terrestrial plants and may achieve biomass productivities severalfold greater than conventional crops because of their simple cellular structure and rapid division rates [16]. Microalgae also have relatively high surface-area-to-volume ratios and efficient carbon assimilation pathways that allow for rapid uptake of inorganic carbon species, supporting elevated carbon fixation rates during cultivation [17]. Another advantage of algal systems is their compatibility with integrated resource recovery frameworks. Microalgae can assimilate nitrogen and phosphorus directly into cellular biomass while removing contaminants from wastewater streams, allowing cultivation systems to simultaneously address atmospheric carbon removal, wastewater treatment, nutrient cycling, and resource recovery within a circular economy framework [18–20].

Despite its efficiency, algae-based carbon capture is often transient; when algae are used for biofuels or feed, the captured CO_2 is eventually re-released, highlighting that biological uptake alone does not guarantee durable carbon sequestration [21,22]. As a result, increasing emphasis has been placed on carbon permanence, which refers to the duration over which removed atmospheric carbon remains stored in a reservoir without significant risk of re-release to the atmosphere [23,24]. To achieve more permanent sequestration, the carbon must be "locked" into a stable form. Thermochemical conversion of algal biomass into biochar through pyrolysis offers one such pathway. Biochar, a carbon-rich solid produced from the pyrolysis of biomass, offers a sequestration horizon of centuries to millennia when produced under optimized conditions and simultaneously provides opportunities for integration with nutrient recovery and soil amendment applications [14,25,26]. Biochar research has historically focused on lignocellulosic feedstocks such as wood and crop residues, and most assessments of biochar as a CDR method are based on these materials [21,22,26]; the long-term carbon stability of algal biochar has not yet been directly established. Reviews of algal biochar have grown in number in recent years but mainly focus on production methods, physicochemical properties, and potential applications such as wastewater treatment, pollutant adsorption, and soil amendment [14,19,27,28]. Few have evaluated the "algae-to-biochar" pipeline as a whole from cultivation and conversion through carbon permanence, life cycle performance, and carbon market requirements. This gap matters because algal feedstocks differ from their terrestrial counterparts in cultivation requirements, biochemistry, byproducts, energy requirements, and costs, in ways that affect each stage of the pipeline. Therefore, findings from lignocellulosic biochar cannot simply be applied to algae. This review explores the optimization of this algae-to-biochar pipeline as a viable global carbon sequestration strategy. Emphasis is placed on advances in cultivation systems, strain engineering, thermochemical conversion pathways, integrated frameworks, and environmental and economic assessments including Life Cycle Assessment (LCA) and Techno-Economic Analysis (TEA). By synthesizing recent developments across these areas, this review aims to assess the potential of integrated algae-to-biochar systems as practical contributors to long-term global carbon management strategies.

This review follows the algae-to-biochar pipeline shown in Figure 1. Section 2 examines strain selection and cultivation systems, and Section 3 covers pyrolysis and how algal biochar compares with terrestrial biochar. Section 4 considers integrated biorefinery frameworks and biochar permanence, and Section 5 reviews life cycle, techno-economic, and policy assessments. Finally, Section 6 discusses barriers to large-scale deployment and priorities for future research. Throughout, microalgal and macroalgal feedstocks are distinguished where their

properties differ, and the origin of the CO_2 supplied to cultivation (atmospheric, biogenic or fossil) is specified, because only atmospheric or sustainably sourced biogenic CO_2 can support a removal claim.

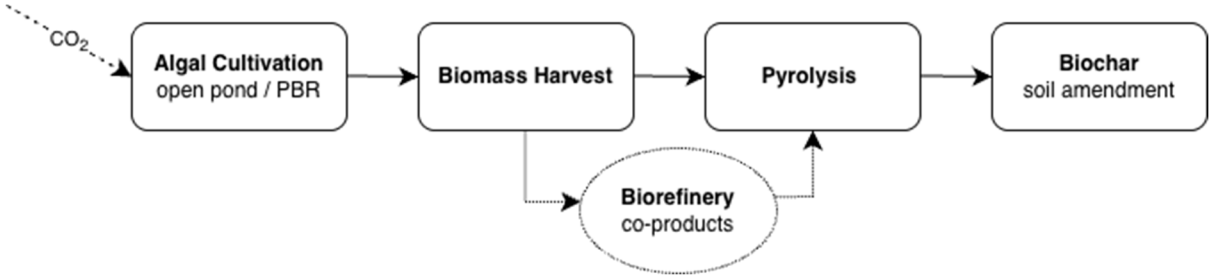


Figure 1. Integrated algae-to-biochar pipeline for net-negative carbon sequestration.

2. Optimized Algal Systems for Enhanced Carbon Capture

Algae grow significantly faster than terrestrial plants and can be cultivated in fresh, saline, brackish, and wastewater environments, meaning algal systems need not compete with food production for arable land and freshwater resources [14]. This combination of rapid growth and resource flexibility makes algal systems particularly attractive for carbon capture at scale [13]. Compared to land-based vegetation, algae can achieve significantly higher carbon fixation rates per unit of land and water because of their faster cell division, direct access to dissolved inorganic carbon (DIC), and capacity for continuous cultivation under controlled conditions [13,18,29].

Recent work has highlighted the importance of both biological characteristics and engineered delivery in realizing the carbon capture potential of algae. In natural systems, the fate of fixed carbon can vary widely based on species and biochemical pathways. Marine algae can invest up to about 80% of the carbon they fix in polysaccharides (glycans), some of which resist bacterial breakdown and store carbon in the ocean over long periods [30]. Fucoid brown algae, for example, export carbon-rich fucoidan compounds into deeper waters, effectively transferring biologically fixed carbon away from rapid surface recycling [31]. Green macroalgae such as *Ulva* produce ulvan, another polysaccharide that can account for roughly 13–39% of dry weight in *Ulva* species, suggesting a similar potential for long-term storage [32,33]. Large-scale seaweed cultivation builds on these natural pathways by expanding macroalgal growth across coastal and offshore waters and has been proposed as a promising CDR strategy because of its large removal potential and relatively low cost [15]. Engineered systems can extend this potential even further beyond the aquatic environments where algae naturally occur. By housing algae in designed systems, carbon capture can be brought to locations such as cities and industrial sites, where natural algal populations are largely absent. Microalgae are particularly well suited to this approach, and researchers are exploring how integrating them into built environments can augment urban carbon capture while providing co-benefits. Microalgae-integrated building enclosures, for example, couple photosynthetic carbon uptake with air-quality improvement and passive shading, creating multifunctional surfaces that simultaneously capture carbon from ambient air and reduce building thermal loads [34].

How much carbon an algal system captures, and whether that carbon stays captured, depends on several interacting variables. Light, temperature, nutrients, and CO_2 supply all drive growth and photosynthetic activity, while the fate of the harvested biomass determines whether the fixed carbon ends up durably stored or returned to the atmosphere [35,36]. Light and temperature both have an optimal range for algal growth and activity. Peak photosynthetic rates have been reported at light intensities between 200 and 400 μmol photons m^{-2} s^{-1} and temperatures between 20 and 30 °C, beyond which photoinhibition and thermal stress reduce productivity, respectively [37,38]. Nutrient availability follows a similar optimization pattern with *Chlorella vulgaris* (*C. vulgaris*) cultures achieving peak biomass productivities above 0.36 g L^{-1} with moderate nitrate and phosphate supplementation, whereas both nutrient-limited and nutrient-excess conditions yielded lower productivities [38,39]. CO_2 supply is also a significant lever. Controlled supplementation through sparging or bicarbonate has been shown to increase algal biovolume by 200–600% and raise areal biomass productivities to peaks exceeding 20 g m^{-2} d^{-1} [40,41].

The origin of the CO_2 supplied to these systems determines whether they remove carbon from the atmosphere or only reduce emissions, and three sources should be distinguished. Atmospheric CO_2, taken up from ambient air or from air-equilibrated water, is the only source that yields atmospheric removal in the strict sense. Biogenic CO_2, for example from fermentation, anaerobic digestion or biomass combustion, was recently withdrawn from the atmosphere, so capturing it and storing it as biochar can also count as removal if the upstream biomass is sustainably sourced. Fossil CO_2 from power plant or industrial flue gas, by contrast, was never part of the atmospheric pool; converting it to biochar can reduce or delay the emissions of that facility, but it does not by

itself remove CO_2 from the atmosphere and should not be counted as a negative emission [7]. The distinction matters because most of the high productivities reported for algal systems rely on concentrated CO_2 supplied by sparging or flue gas [40,41]. Cultivation on ambient air is much more constrained. Atmospheric CO_2 is only about 0.04% by volume (roughly 422 ppm [4]), so the driving force for gas-to-liquid transfer is small and dissolved CO_2 is quickly depleted in dense, actively photosynthesizing cultures. Air-fed systems therefore depend on bicarbonate uptake through CCMs, large gas–liquid contact areas or intensive aeration, and the energy needed to move large volumes of air can offset part of the carbon captured [42–44]. For example, a combined techno-economic and life cycle assessment of an algae-based coating that captures CO_2 directly from air and converts the harvested biomass to biochar estimated net carbon removal efficiencies of only 27–49% and costs of US$702–1585 per tonne of CO_2 net sequestered, with pyrolysis exhaust gas as the largest emission source [45]. In this review, net-negative claims are therefore limited to systems in which the carbon stored in biochar originated from atmospheric or sustainably sourced biogenic CO_2, and even then only where a full life cycle assessment shows that emissions across the process chain do not exceed the carbon stored.

2.1. Strain Selection and Genetic Optimization

The selection of an appropriate algal strain is the foundational determinant for optimizing carbon capture performance because biochemical characteristics vary substantially among taxa, directly influencing growth rate, biomass productivity, nutrient removal behavior, stress tolerance, and biomass composition [46,47]. These differences lead to large variations in productivity under otherwise identical conditions. For example, among ten strains of *Chlorella* cultivated in municipal wastewater influent, the CO_2 fixation efficiency of four strains was significantly higher than that of the others [46]. Another comparative cultivation of three green microalgae strains showed that *Chlorella sorokiniana* achieved the highest biomass concentration, specific productivity, and CO_2 utilization efficiency compared with *Neochlorella semenenkoi* and Desmodesmus armatus [48]. Beyond simple productivity metrics, strain selection should also consider metabolic flexibility and nutrient utilization efficiency to ensure long-term operational stability in engineered systems. Genome-scale metabolic modeling of *C. vulgaris* under light-dark cycles demonstrated that optimized nutrient supply strategies could reduce glucose demand by ~75% and nitrate inputs by ~23% while maintaining more than 80% of the biomass productivity observed in glucose-fed heterotrophic cultures [39]. These findings highlight how effective strain selection serves as the foundation for downstream optimization. Once suitable strains have been identified, genetic optimization strategies can be applied to further enhance carbon fixation efficiency.

2.1.1. RuBisCO Engineering

One of the most widely studied targets is the CO_2-fixing enzyme ribulose-1,5-bisphosphate carboxylase/oxygenase, RuBisCO. RuBisCO catalyzes the first and most critical step of the Calvin cycle, the set of reactions through which algae and plants convert atmospheric CO_2 into organic compounds during photosynthesis [49]. However, RuBisCO is notably slow and prone to a competing reaction with O_2 rather than CO_2, a process called oxygenation, which reduces net carbon fixation efficiency [50,51]. One practical route to indirectly improve RuBisCO function is to enhance its activation state. In *Nannochloropsis oceanica,* a candidate RuBisCO activase was transcriptionally induced under low CO_2 and increased growth rate by ~32% and biomass productivity by ~46% [52]. Protein engineering approaches such as directed evolution and site-directed mutagenesis target the enzyme's active site to improve its selectivity for CO_2 over O_2, thereby reducing efficiency losses due to oxygenation [50,53]. Heterologous expression strategies take a different approach to reducing oxygenation by introducing RuBisCO variants from other organisms that naturally exhibit higher CO_2 specificity than the native form. One potential RuBisCO variant isolated from *Griffithsia monilis* has a CO_2/O_2 specificity factor of 167, two- to fourfold higher than that of typical green algae [49,54].

2.1.2. Carbon Concentrating Mechanisms (CCMs)

Complementary strategies to RuBisCO engineering enhance intracellular carbon availability through CCMs. CCMs work by actively transporting inorganic carbon into the cell, thereby raising local CO_2 concentrations well above ambient levels and suppressing the competing oxygenation reaction [55]. Many microalgae operate CCMs that actively transport bicarbonate into the cell and concentrate CO_2 around RuBisCO within pyrenoids, specialized microcompartments that mediate ~28–44% of global carbon fixation [56,57]. The primary targets for CCM optimization are overexpression of bicarbonate transporters and carbonic anhydrase, both of which increase the rate and efficiency of CO_2 delivery to RuBisCO [49]. However, many microalgae downregulate CCM components as external CO_2 concentrations rise, which creates a conflict in algae-to-biochar systems that supplement CO_2 [50].

For example, flue-gas-fed cultivation is attractive because it boosts biomass productivity while reusing an industrial waste stream [18,40]. Yet flue gas supplies CO_2 well above concentrations at which CCMs are downregulated, so engineered CCM improvements may offer little benefit under these conditions [40,50]. Conversely, CCM engineering is likely to be most valuable in systems that rely on atmospheric CO_2, where low dissolved CO_2 concentrations limit carbon fixation, which is also the case in which the stored carbon qualifies as removal.

2.1.3. Tailoring Biomass Composition

Controlling the composition of algal biomass is another key optimization strategy, in particular the balance between lipids and carbohydrates. The lipid-rich mutant *Chlamydomonas* sp. KOR1 was developed through light/dark-conditioned screening, which disrupted starch remodeling and accelerated repartitioning of carbohydrates into lipids, demonstrating that carbon allocation can be steered rather than treated as a fixed trait [58]. Similarly, *Synechococcus elongatus* was metabolically engineered to produce fatty acid ethyl esters (FAEEs) by expressing a heterologous ethanol-producing pathway and a wax ester synthase [59]. Since thermochemical behavior depends on biochemical composition, tuning the lipid-to-carbohydrate ratio provides a direct link between metabolic engineering and downstream biochar yield and quality.

Most composition engineering, however, has targeted lipid accumulation for biofuels, and this goal is not necessarily aligned with biochar production. Lipids decompose mainly between 350 °C and 550 °C (Section 3) and leave the reactor largely as aliphatic vapors that condense into bio-oil rather than char [27]. Raising cellular lipid content would therefore be expected to shift pyrolysis products toward bio-oil and lower the share of feedstock carbon retained in biochar, whereas carbohydrates and proteins contribute more directly to primary char formation [27]. Consistent with this, pyrolysis of lipid-extracted microalgal residues yields relatively more nitrogen-containing compounds and fewer aliphatic hydrocarbons than pyrolysis of the raw biomass [60]. For algae-to-biochar systems, strain selection and engineering should therefore also target traits that govern char yield and quality: ash content and mineral composition, which add non-carbon mass to the char and can influence carbon retention [61,62]; nitrogen content, which affects both nutrient value and the release of NO_x precursors during pyrolysis (Section 3.2) [63]; and cell wall composition, which differs widely among algal taxa and has received little attention as a determinant of char yield. Where lipids are the engineered product, they are better treated as a co-product extracted before pyrolysis (Section 4.1) than as a contributor to biochar yield.

Taken together, these strategies illustrate that maximizing carbon capture in algal systems requires a multilevel approach that begins with selecting appropriate strains and extends through targeted metabolic optimization. By combining these efforts, it becomes possible to significantly enhance both carbon fixation efficiency and downstream sequestration potential of algal biomass. However, most of these genetic strategies have been demonstrated only in laboratory cultures. Engineered traits may be lost or altered outdoors as cells encounter native competitors and fluctuating conditions. In the first EPA-sanctioned outdoor trial of an engineered alga, *Acutodesmus dimorphus* retained its introduced traits for 50 days of open pond cultivation, but engineered cells dispersed from the ponds [64]. Biocontainment strategies to limit the survival of escaped cells have mostly been tested under controlled conditions and outdoor cultivation of engineered strains requires regulatory approval, such as from the EPA in the United States [65]. These barriers make closed photobioreactors more practical for engineered strains and limit use in open systems, both of which are discussed in Section 2.3.

2.2. Cultivation Systems

Even the best-optimized strain will underperform in a poorly designed system. At scale, the cultivation environment itself often becomes the primary bottleneck [13,35]. In high-density cultures, self-shading creates steep light gradients, leading to simultaneous light limitation and photoinhibition within the same system. Mechanistic modeling has demonstrated that suboptimal light distribution within cultivation systems can reduce overall productivity even when sufficient nutrients and CO_2 are available [66]. Similarly, insufficient gas transfer can constrain carbon fixation even when light is abundant; however, excessive mixing and/or sparging increases energy demand and reduces overall system efficiency [41,67]. Additional factors such as temperature fluctuations, contamination risk, and water availability also significantly affect long-term performance of large-scale algal systems [68,69]. In short, cultivation performance is fundamentally a function of system design and different reactor configurations impose distinct trade-offs between efficiency, cost, and operational complexity. Most large-scale algal systems fall into two broad categories: open systems and closed systems. The following section compares these systems in detail, focusing on their implications for carbon capture efficiency.

2.3. Ponds vs. Photobioreactors (PBRs)

Open ponds and closed PBRs represent the two dominant approaches to large-scale microalgal cultivation. These approaches differ not only in their engineered design but also in how they address environmental variability, scalability, and cost [70]. Both open and closed systems have demonstrated potential for large-scale microalgal cultivation, but for systems targeting long-term carbon sequestration rather than solely biomass generation, these engineering differences become increasingly important. Typical examples of open and closed algal cultivation systems are shown below in Figure 2.

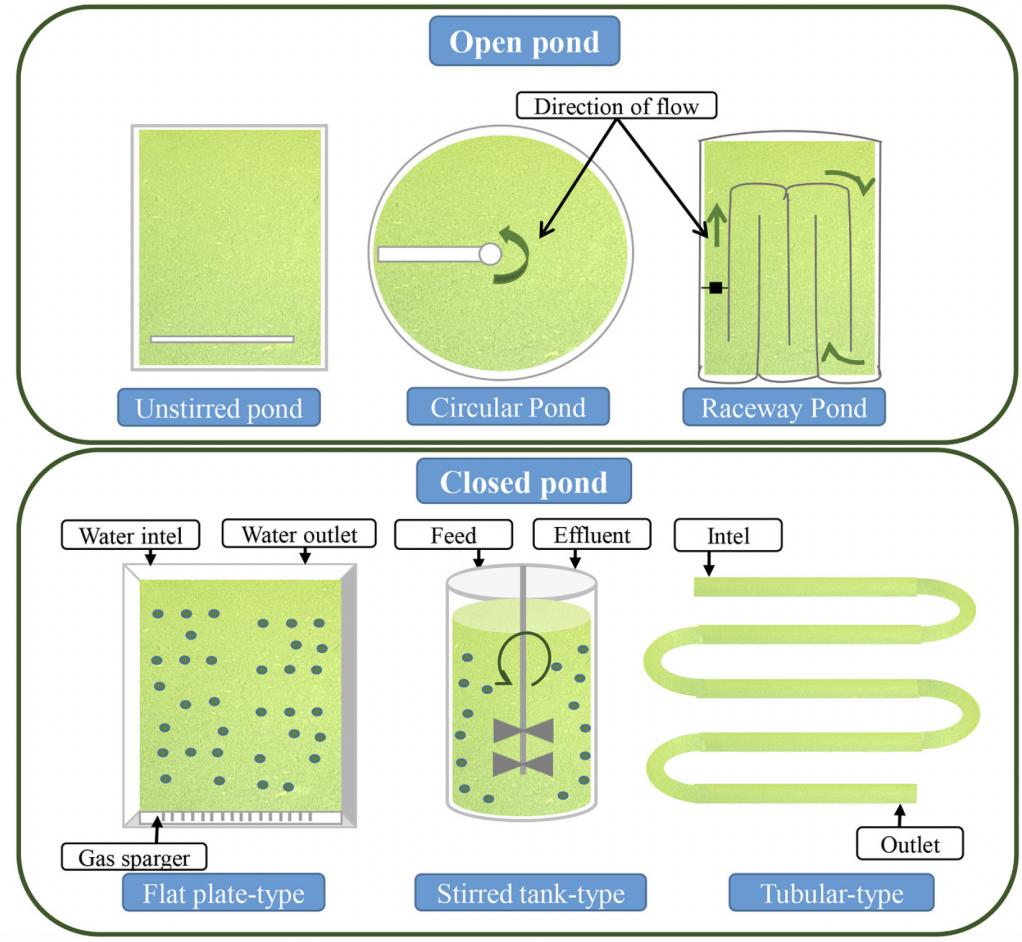


Figure 2. Schematic representation of common open and closed algal cultivation systems. Reprinted with permission from ref. [71], under CC BY 4.0.

Open ponds provide a major economic advantage because of their relatively low construction and operating costs, as well as their ability to be deployed at large scale [72]. However, environmental exposure in outdoor systems introduces variability. Temperature fluctuations, both diurnal and seasonal, can push cultures outside their optimal growth range and reduce system efficiency [68]. Contamination is an equally persistent issue, as open ponds are vulnerable to invasion by competing organisms that can collapse a culture and require ongoing monitoring and intervention to control [69]. The lower degree of environmental control in open systems often results in reduced biomass productivity and carbon conversion efficiency [72,73]. For algae-to-biochar systems, this exposure also makes the feedstock more variable as competing species and seasonal changes shift biomass composition. Open ponds also produce dilute cultures, increasing the energy needed for dewatering before pyrolysis, which is already one of the most energy-intensive steps.

In contrast, closed PBR systems provide substantially greater control over environmental conditions and generally achieve higher biomass productivity and carbon conversion efficiency [72,74]. Recent studies demonstrate that optimized PBR designs can significantly enhance CO_2 utilization through improved gas transfer and mixing strategies [75,76]. One study in 2020 reported CO_2 utilization efficiencies approaching 35% through bubble-size optimization in *C. vulgaris,* accompanied by a CO_2 fixation rate of 633.7 mg L^{-1} d^{-1} and biomass concentrations exceeding 4.2 g L^{-1} [75]. Another study in 2021 developed a trickle-bed reactor that utilized immobilized algae to achieve approximately 40% CO_2 removal from flue and exhaust gases at a feed rate of 25 dm^3 h^{-1} [77]. CO_2 removal

efficiencies exceeding 90% have even been achieved by optimizing gas sparging and mass transfer kinetics [42,76]. These findings demonstrate that PBRs are well-suited for potentially net-negative systems where maximizing carbon retention through downstream conversion is the priority. The controlled conditions of PBRs make them a strong candidate for algae-to-biochar systems as they yield a more consistent feedstock and thus more predictable biochar properties. In addition, the higher culture densities typical of PBRs reduce the volume of water that must be removed before pyrolysis. However, translating these performance advantages to large-scale deployment introduces a new set of challenges.

PBR deployment remains constrained by increased capital investment and operational energy requirements associated with reactor construction, pumping, mixing, and various environmental controls [70,78]. Therefore, the productivity advantages of closed systems must be weighed against their greater economic burden. A comparative productivity and techno-economic analysis of open raceway ponds and flat-panel PBRs demonstrated that PBRs achieved substantially greater biomass productivities, 3800–13,000 t km^{-2} $year^{-1}$, than open raceway systems, 2000–7200 t km^{-2} $year^{-1}$ [72]. However, these productivity gains came at substantially greater cost, with open systems exhibiting capital and operational cost values of approximately \$131–341 t^{-1} and \$408–434 t^{-1}, respectively, compared to \$876 t^{-1} and \$2576 t^{-1}, respectively, for flat-panel PBRs [72]. This emphasizes the persistent trade-off between enhanced productivity and economic scalability in microalgal carbon capture systems.

Consequently, hybrid approaches have increasingly emerged as a practical compromise, employing PBRs during early cultivation stages to establish stable, high-density cultures before transferring biomass to lower-cost open systems for large-scale growth [79]. Such approaches may combine the productivity advantages of controlled systems with the economic feasibility of open cultivation, but they can also inherit vulnerabilities from both (Table 1). For example, hybrid systems reduce contamination risk compared to open ponds by limiting how long cultures are exposed, since PBRs maintain a high-density inoculum that allows the open stage to run as short batch cycles [79]. However, the open stage remains vulnerable to grazers that can collapse a culture within days [69]. For biochar production, hybrids inherit the dilute harvest and greater variability of open ponds since their final growth stage takes place there, though the PBR stage provides a more consistent starting culture.

It is worth noting that the productivity and cost figures reported throughout this review should be interpreted with caution. Reported biomass yields and costs vary widely across studies because they depend on location, climate, algal species, nutrient source, and the scale at which results were obtained. Even within the comparative analysis between open ponds and flat-panel PBRs, productivities span roughly threefold for each system [72]. Furthermore, a global assessment of a single cultivation design across 6685 locations found that productivity, costs, and environmental impact all varied substantially with location [80]. These values are therefore best understood as condition-specific ranges and individual point values should not be extrapolated directly to other sites, species, or scales. For this reason, Table 1 presents qualitative comparisons of open, closed, and hybrid systems rather than specific values.

Table 1. Comparison of open (ponds), closed (PBRs), and hybrid algal cultivation systems.

Parameter	Open Pond	Photobioreactor	Hybrid System
Biomass productivity	Low	High	High
Capital cost	Low	High	High
Operating cost	Low	High	Moderate
Land requirement	High	Low	High
Contamination risk	High	Low	Moderate
Temperature control	Low	High	Moderate
Evaporative losses	High	Low	Moderate
Operational complexity	Low	High	High
Scalability	Easy	Difficult	Complex

3. Thermochemical Conversion from Algal Biomass to Biochar

In response to urgent global goals for climate change mitigation and sustainable resource management, substantial scientific interest has been redirected to NETs. Among the diverse assortment of NETs is the thermochemical conversion of biomass into biochar (Figure 3), a persistent carbon-rich solid residue and a leading option for the removal and storage of atmospheric CO_2 [25].

Historically, lignocellulosic biomass has dominated the biochar research arena; however, algal (both micro- and macroalgae) biomass has recently emerged as a promising third-generation feedstock given its intrinsic photosynthetic efficiency, rapid growth rates, and adaptability to a wide range of media [27]. The "net-negative" nature of the algal biochar is not an intrinsic property but is instead governed by the complex variables of the

pyrolysis process. Pyrolysis is defined as the thermal decomposition of organic materials in an oxygen-limited environment and typically occurs in multiple stages [81]. The first stage of the thermal degradation of biomass occurs between 25 °C and 200 °C and primarily involves the loss of moisture and evaporation of lightly bound volatile compounds [27]. Increasing temperatures result in the primary degradation of carbohydrate and protein components between 200 °C and 500 °C, with the dissolution and decomposition of lipid macromolecules occurring between 350 °C and 550 °C (Table 2) [27].

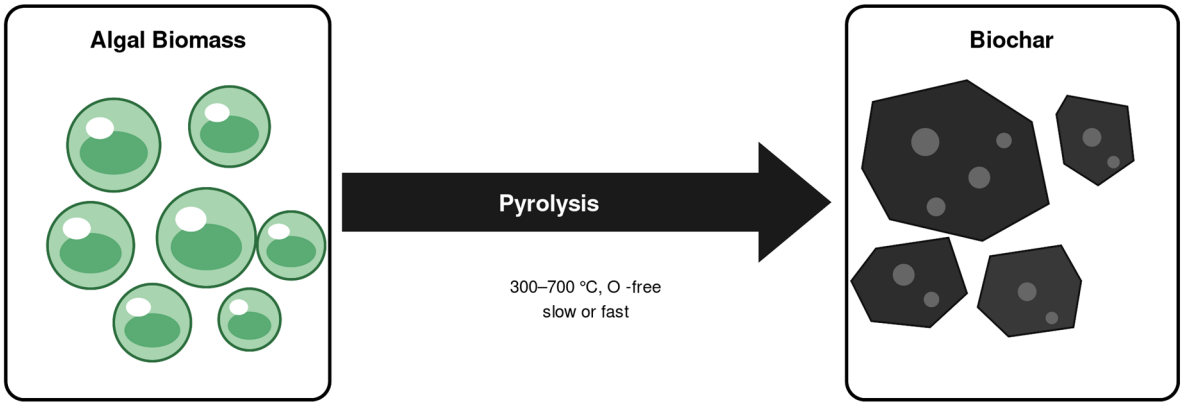


Figure 3. Thermochemical conversion from algal biomass to biochar via pyrolysis.

Microalgae and macroalgae should not, however, be treated as a single feedstock class. Microalgal biomass is typically rich in protein and, under nutrient stress, in lipid, and the whole cell is pyrolyzed [63]. Macroalgal biomass is generally dominated by structural and storage polysaccharides such as alginate, ulvan, agar and carrageenan and contains comparatively little lipid [62]. Marine macroalgae also tend to carry high loads of salts and other minerals, so their biochars are typically low in carbon and surface area but high in ash, pH and extractable nutrients [62]. Much of the published algal biochar characterization data, including several studies on *Ulva*, *Sargassum* and *Cladophora* cited in this section, comes from macroalgae, whereas the cultivation and engineering advances in Section 2 concern mainly microalgae. Results from one group should therefore not be transferred to the other without caution. To make this explicit, the feedstock type is identified for the studies discussed below, and the captions of Tables 3–5 indicate whether values derive from macroalgal, microalgal or non-algal feedstocks.

Table 2. Degradation ranges of biochar feedstock components and their subsequent products.

Feedstock Component	Degradation Range (°C)	Principal Products	Reference
Moisture & light volatiles	25–200	Water vapor, low-molecular-weight gases	[27]
Carbohydrates & proteins	200–500	Bio-oil vapors, primary char	
Lipids	350–550	Aliphatic hydrocarbons	
Residual organics & lignin	550–800	Secondary char, syngas (H_2, CO)	

Table 3. Attributes of biochars produced at different pyrolysis temperatures. Values are indicative ranges compiled mainly from non-algal biochars (lignocellulosic and spent mushroom substrate) [61,82] and are not specific to algal feedstocks; mean residence times (MRT) are model-based estimates. Lower H/C ratios indicate higher aromaticity.

Pyrolysis Temperature (°C)	Yield (wt%)	Carbon Stability (MRT in Years)	Aromaticity Indicator (H/C)	Reference
300	45–65	90–150	>0.8 (low aromaticity)	[61]
500	25–40	500–800	~0.5 (moderate aromaticity)	
700	15–30	1000–1600	<0.3 (high aromaticity)	[82]

Table 4. Attributes of biochar produced at different heating rates. Values are indicative and compiled from mixed feedstocks, including red macroalgae [83] and non-algal biomass [84]; SSA and pore volume decrease as heating rate increases.

Heating Rate (°C/min)	SSA (BET) Range (m^2/g)	Total Pore Volume (cm^3/g)	Morphology	Reference
5–15 (Slow)	150–350	0.10–0.20	Preserved cellular, high microporosity	[84]
30–50 (Intermediate)	20–100	0.02–0.05	Partially collapsed, mesoporous	[83]
>100 (Fast)	<10	<0.01	Fragmented, low surface area	[27]

Table 5. Comparison of algal and terrestrial biochars. Algal values derive mainly from macroalgal feedstocks unless stated; algal-specific stability data remain scarce.

Aspect	Algal Biochar	Terrestrial Biochar (e.g., Wood)	Sources
Carbon stability	Uncertain; few direct measurements. Low lignin and high ash and N contents; H/C-based estimates can be inflated by carbonate carbon (Section 4.2)	Better documented; centennial-scale persistence in soil (stable-pool MRT of ~556 years in a meta-analysis)	[62,85,86]
Nutrient Content (N/P)	High (5–10% N, 1–3% P; microalgal)	Low (<1% N/P)	[14]
Carbon and ash content	Low carbon, high ash (17–38% ash in algal and seagrass chars)	High carbon, low ash	[62,87]
Char mass yield at 400–600 °C	22–48 wt% (raised by ash)	20–27 wt% (pine)	[87]

3.1. Pyrolysis Parameters

3.1.1. Temperature

Pyrolysis temperature is considered the most significant variable influencing the chemical composition and environmental permanence of the biochar [27]. The pyrolysis temperature range of 500–700 °C represents a critical window where biochar transitions from a relatively labile, aliphatic-dominated residue to a recalcitrant, aromatic carbon sink (Table 3) [82]. At lower temperatures, incomplete carbonization results in a higher content of oxygen functional groups and aliphatic structures [27]. As temperature rises, the rates of dehydration, decarboxylation, and aromatization accelerate, resulting in a more condensed carbon structure [81]. Aromaticity is generally expressed as atomic ratios of H/C and O/C. The decrease in these ratios indicates increased aromatization and decreased polarity [61]. For example, the H/C ratio of hemp-stalk biochars (kenaf, jute and ramie; a lignocellulosic feedstock) decreases as the pyrolysis temperature rises to 700 °C, indicating the formation of more stable graphitic layers [88]. Raman spectroscopy also supports this trend; the I_D/I_G ratio (the ratio of the disorderly D to the graphitic G peak) increases with temperature, indicative of a more ordered (although still largely amorphous) carbon structure [27]. Similarly, in work on the macroalgae *Ulva expansa* and *Sargassum* sp., higher temperatures have been shown to increase the concentration of inorganic minerals as the organic fraction is broken down and the production of sp^2-hybridized carbon, a signature of chemical stability [89]. The environmental implications of this thermal processing are significant. High-temperature (>550 °C) biochars have been found to mineralize much more slowly when added to soil, with mean residence times (MRT) of several centuries to a thousand years [17], although these estimates come mainly from non-algal biochars and have not yet been confirmed for algal biochar. It is this resistance to microbial breakdown that underpins the “net-negative” potential. Additionally, higher temperatures favor the production of mineral-carbon complexes. The addition of exogenous minerals, such as Fe, has been shown to increase carbon retention by up to 44.5% and graphitization at 600 °C in spent mushroom substrate biochar, protecting the carbon from oxidative loss [61]. These gains in stability come at the cost of yield, creating a central trade-off for carbon accounting. Higher temperatures produce less biochar per tonne of biomass, but the remaining carbon is more aromatic and resistant to decay [82]. Net carbon removal therefore depends on both the fraction of feedstock carbon retained in the char and the fraction of that carbon that persists, so the optimal temperature depends on the permanence requirements being targeted [22]. For high-ash algal feedstocks, carbon yield is a more meaningful basis for comparison than mass yield, since a growing share of the char mass is mineral rather than carbon at higher temperatures [89].

3.1.2. Heating Rate

The heating rate determines the physical morphology of biochar [27]. Within the slow pyrolysis heating-rate range of 0.1–20 °C/min, the production of biochar is favored as the slow heating allows sufficient time for secondary repolymerization and recombination reactions [81]. The heating rate influences the mechanism of volatile escape from the algae, with a slow rate resulting in the gradual evaporation of moisture and gases within the algae, preserving the internal cellular structure and enabling the formation of a complex network of pores [28]. The specific surface area (SSA) and pore volume are key parameters that determine the performance of biochar as an adsorbent or a soil amendment. In general, SSA and pore volume decline as the heating rate increases beyond the slow pyrolysis regime, consistent with the progressively collapsed and fragmented morphologies summarized in Table 4 [27,83,84,90]. Pore development depends more strongly on temperature, because higher temperatures

remove more completely the volatiles that would otherwise occlude the pores [27]. Scanning Electron Microscopy (SEM) images of algal biochars (mostly from macroalgae) typically reveal a more compact structure than that of wood-derived chars, with macro- and micropores partially filled or lined with salt crystals and carbonates [89,91]. This reflects the higher inorganic mineral content characteristic of algal feedstocks, as illustrated in Figure 4.

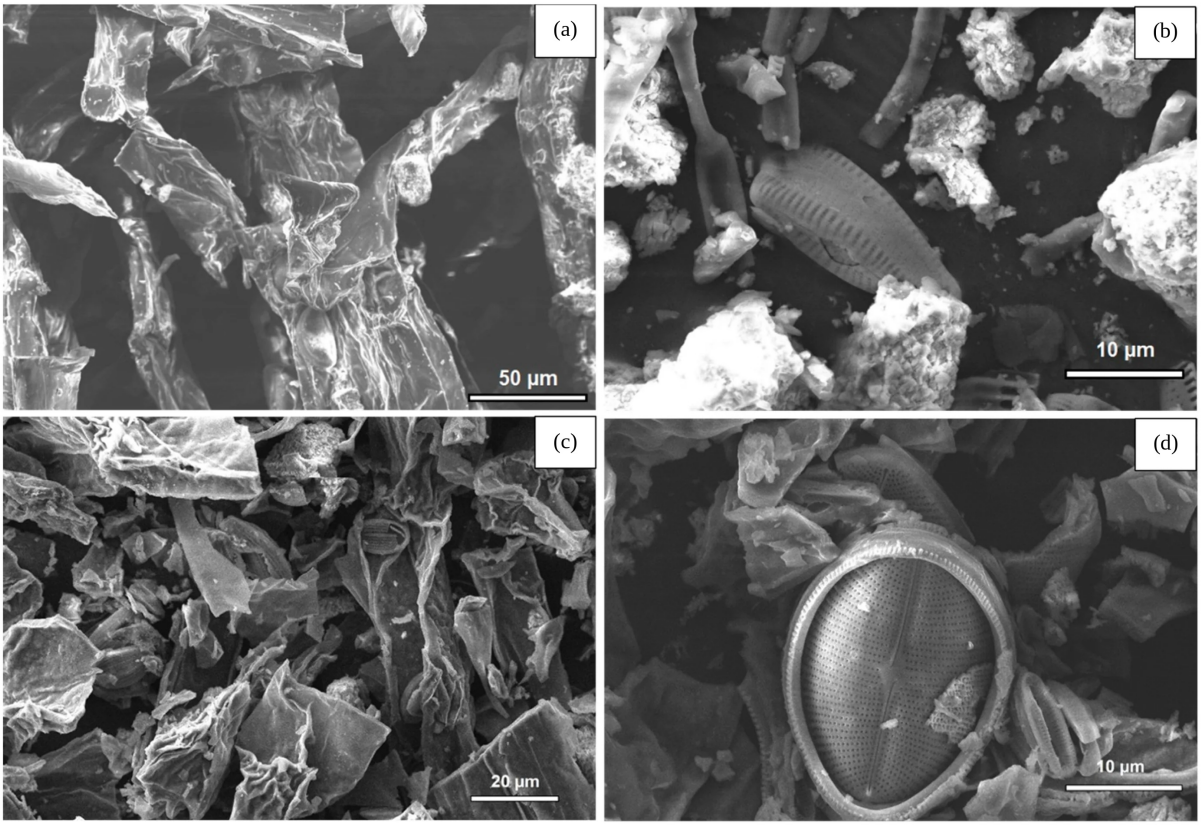


Figure 4. SEM images of (**a**,**b**) raw Cladophora glomerata and (**c**,**d**) biochar obtained at 450 °C. Reprinted with permission from ref. [92], under CC BY 4.0.

The heating rate affects the kinetic behavior of the biomass during heating. Thermogravimetric analysis (TGA) indicates that most combustion and decomposition parameters increase with heating rate, whereas the residence time at a given temperature decreases [93]. This "thermal lag" may result in incomplete carbonization if the residence time is not adjusted accordingly. The activation energy for the decomposition of algal biomass is generally higher than that of lignocellulosic residues, indicating more complex, and therefore more stable, molecular structures involved, especially for protein-rich species [93]. Thus, a slower heating rate is required to produce high-surface-area biochar for environmental remediation, as it allows uniform thermal penetration and unimpeded development of micropores [89].

Most of the property data summarized in Tables 2–4, whether for algal or non-algal feedstocks, come from small-scale laboratory pyrolysis, typically in batch furnaces or thermogravimetric analyzers that process gram quantities of material under tightly controlled conditions. Industrial-scale reactors differ in heat transfer, vapor residence time, and temperature uniformity, so yields and stability measured in the laboratory may not carry over directly. In one recent comparison, biochar yield fell from an average of 37.1% in a laboratory furnace to approximately 26% in a pilot-scale reactor using the same feedstock (cotton gin waste), a difference the authors attributed largely to air entering the pilot rotary kiln, while carbon stability remained high at both scales [94]. Comparable lab-to-pilot comparisons have not yet been reported for algal feedstocks, so the values in Tables 2–4 should be viewed as laboratory benchmarks until they are confirmed at larger scales.

3.2. Comparison of Algal and Terrestrial Biochar

It is essential to understand the underlying biochemical processes to produce the maximum fraction of solid biochar. Algal biochar typically contains higher nitrogen and phosphorus content compared to terrestrial, wood-based biochar [89,95]. This makes it not only a carbon sink but also a potent "bio-fertilizer," facilitating nutrient circularity in agriculture and reducing reliance on synthetic fertilizers [96]. However, the same nitrogen that gives

algal biochar its fertilizer value also creates emission risks during pyrolysis. Protein-bound nitrogen in algae is partly released as ammonia (NH_3) and hydrogen cyanide (HCN), both precursors to nitrogen oxides (NO_x), or condenses into nitrogen-containing heterocyclic compounds [63]. If pyrolysis gases or bio-oil are burned to supply process heat, these compounds can be converted to NO_x, so algae-to-biochar facilities may need emission controls that are less of a concern for low-nitrogen terrestrial feedstocks. Catalytic approaches may help reduce this risk. For example, adding CaO during microalgae (*Nannochloropsis gaditana*) pyrolysis retained 41.1% of feedstock nitrogen in the char and reduced the release of NH_3 and HCN [97]. Terrestrial (i.e., lignocellulosic) biomass is dominated by cellulose, hemicellulose, and lignin, while algal biomass is composed of a heterogeneous matrix of carbohydrate, protein, and lipid macromolecules [93]. Part of the fundamental difference between terrestrial and algal biomass includes the lack of lignin in many algal species. Lignin is the component that primarily contributes to the structural stability and high carbon yields of woody biochar [83]. This, however, is not the case for algae because of the high protein content, which brings nitrogenous compounds into the carbon matrix. In the Maillard reaction, the interaction between reducing sugars and amino acids during pyrolysis leads to the conversion of carbonyl compounds into nitrogenous heterocycles with higher thermal stability [98]. This indicates that a distinct chemical recalcitrance and surface functionality may arise in algal biochar compared to conventional, wood-derived biochar [19]. The absence of lignin would also be expected to reduce char yields, yet reported mass yields of algal biochar are often comparable to or higher than those of woody feedstocks. In one direct comparison at 400–600 °C, a freshwater alga and the seagrass *Posidonia oceanica* yielded 22–48 wt% char compared with 20–27 wt% for pine, while containing 17–38% ash (Table 5) [87]. Because this ash adds mineral mass to the char and minerals can also promote carbon retention, carbon yield offers a fairer basis than mass yield for comparing algal and terrestrial biochars [61]. A higher mass yield does not, however, mean more stored carbon. Across eight green-tide macroalgae, biochars were comparatively low in carbon content, surface area and cation exchange capacity but high in ash and nitrogen, and were judged volumetrically less able than high-carbon lignocellulosic biochars to deliver carbon sequestration benefits [62]. Claims that algal biochar is more stable than wood biochar should therefore be treated with caution until organic-carbon-based stability data for algal chars are available (Section 4.2).

3.3. Wet Conversion Alternative: Hydrothermal Carbonization

Because pyrolysis requires low feedstock moisture, the energy needed to dry wet algal biomass can erode much of the carbon benefit (Sections 5.1 and 6). Hydrothermal carbonization (HTC) avoids drying altogether: wet biomass is heated in water under autogenous pressure, typically at about 180–260 °C, to produce a carbon-rich solid (hydrochar) and a nutrient-rich process water [99,100]. Microalgae are well suited to this route. Heilmann et al. converted microalgal slurries into a coal-like char at about 200 °C with reaction times under 1 h and pressures below 2 MPa [101], and hydrochars have also been produced from wastewater-grown algal biomass [19]. The trade-off is permanence. Because HTC operates at much lower temperatures than pyrolysis, hydrochars generally retain more hydrogen- and oxygen-rich functional groups, with higher H/C and O/C ratios and lower expected stability in soil than pyrolysis biochars [100]. Hydrochar is also excluded from some crediting methodologies; VM0044, for example, does not accept hydrothermal carbonization as a biochar production route [102]. HTC may therefore be most useful as a low-energy pretreatment or dewatering step, or where nutrient recovery is the main goal, rather than as a direct substitute for pyrolysis when durable carbon storage is the objective. Related wet routes such as hydrothermal liquefaction mainly produce bio-crude and are reviewed elsewhere [19,99].

4. Integrated Net-Negative Frameworks

For algal carbon capture systems to function as credible net-negative strategies, they must extend beyond biomass production alone and integrate carbon capture, biomass utilization, resource recovery, and durable carbon storage within a unified system boundary [43,103,104]. Even though algal cultivation can achieve rapid biological carbon fixation, long-term climate benefits depend on preventing the captured carbon from being re-released to the atmosphere via degradation. To address this, attention has increasingly shifted toward integrated frameworks that simultaneously optimize environmental performance and economic viability through downstream product recovery and carbon stabilization strategies. Recent studies have demonstrated that these integrated approaches can significantly improve environmental performance of algal systems relative to their isolated counterparts [105,106].

4.1. The "Biorefinery" Approach

A truly optimized system uses a cascading biorefinery approach in which high-value products are extracted prior to final carbon stabilization. Rather than treating the harvested biomass as a single output feedstock, integrated algal biorefineries maximize value by fractionating biomass into multiple value-optimized product

streams prior to carbon stabilization. Most of the evidence for cascading algal biorefineries, however, remains conceptual. In one techno-economic model of an integrated microalgal biofuel biorefinery, co-products such as polyurethane and valorized residual solids were estimated to account for over 80% of total system revenue, with fuel sales contributing as little as 7% [106]. This is a model-dependent estimate rather than an observed outcome: it reflects the product slate, assumed market prices and process design of that specific scenario and has not been demonstrated in an operating biorefinery. This revenue share reflects price rather than mass: specialty co-products command a much higher value per tonne than fuels, which must compete with petroleum on price [106]. The result is therefore sensitive to co-product prices and to whether markets for these products could absorb the volumes implied by large-scale cultivation. Beyond economics, this type of approach improves resource utilization and reduces waste generation. Compounds such as proteins, pigments, lipids, antioxidants, and specialty biologics can be recovered before the remaining residue is converted into biochar (Figure 5) [107].

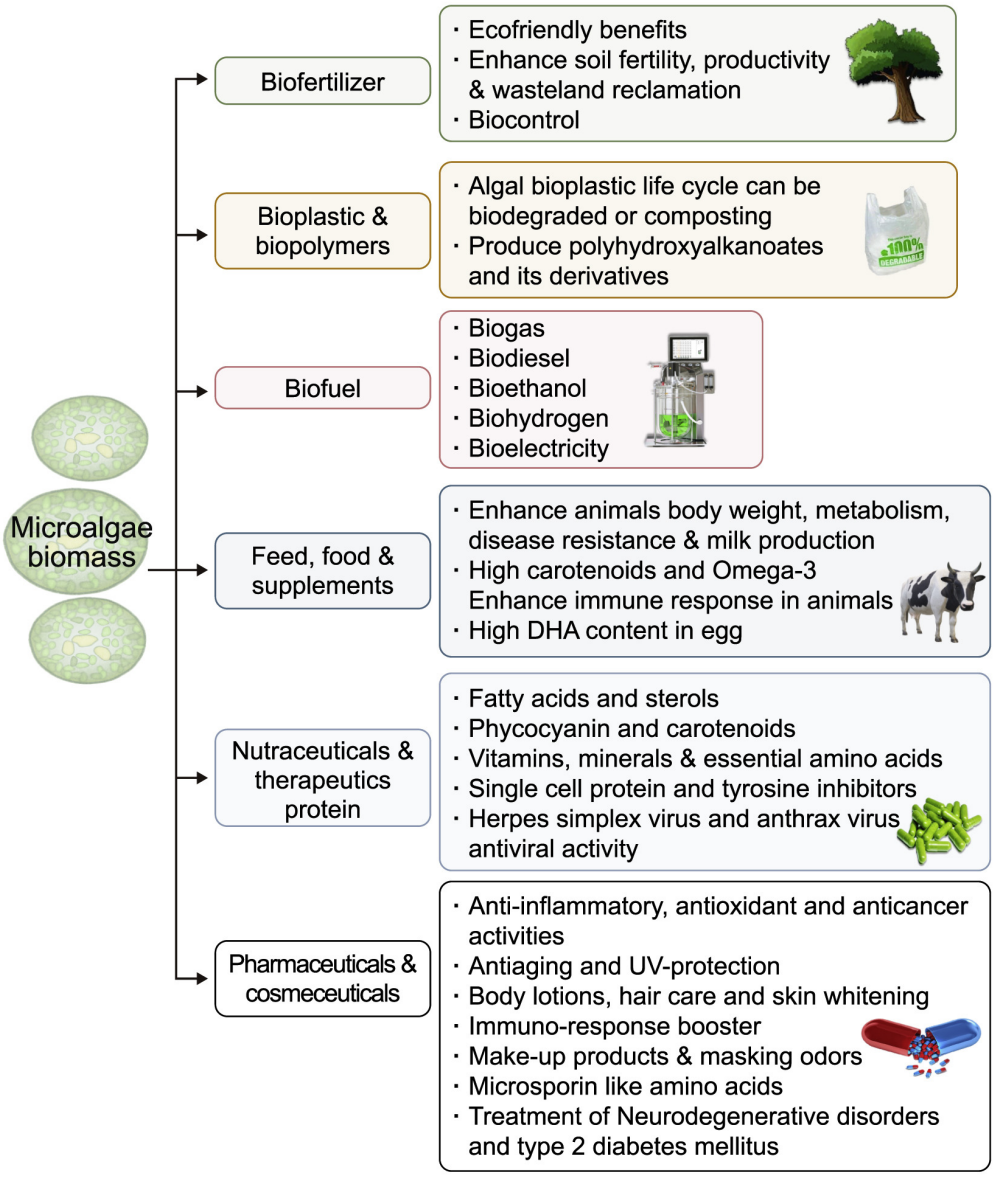


Figure 5. Potential high-value biorefinery products from microalgal biomass before pyrolysis. Reprinted with permission from ref. [108], under CC-BY-NC-ND 4.0.

High-value pigments represent one particularly promising pathway. For example, astaxanthin—a carotenoid with strong antioxidant properties and applications in nutraceutical, pharmaceutical, and cosmetic industries—is widely produced from *Haematococcus pluvialis,* which is recognized as one of the most promising and sustainable sources of this high-value pigment [109–111]. Following extraction of these valuable products, the residual biomass retains significant carbon content and can then be directed toward pyrolysis, allowing the system to simultaneously produce marketable products and durable carbon storage [110]. This cascading framework

transforms algae from a single-purpose carbon-capture feedstock into a multifunctional platform that supports carbon removal, nutrient recovery, and bio-based product generation [107].

Co-product extraction, however, also changes the feedstock that reaches pyrolysis. Removing lipids leaves a residue enriched in protein, carbohydrates, and ash. Pyrolysis of lipid-extracted microalgae has been shown to yield more nitrogen-containing compounds than pyrolysis of the raw biomass [27,60]. A higher yield of nitrogen-containing heterocycles (Section 3.2) can add thermal stability to the resulting biochar but also increases NO_x risk during processing [98]. Meanwhile, a higher ash fraction raises mass yield even where carbon retention changes little, making carbon yield essential for judging the outcome [61]. Because these effects have not yet been measured together, the benefits of pairing co-product recovery with biochar production remain primarily theoretical.

4.2. Carbon Sequestration Potential and Stability

The carbon sequestration value of an algal biochar system depends not only on how much biomass is produced but also on the fraction of that biomass carbon that remains stable after thermal conversion and soil application. This distinction is important because raw, untreated algal biomass is biologically labile and can rapidly return fixed carbon to the atmosphere through respiration, microbial decomposition, or short-lived product use [112]. Pyrolysis changes this fate by converting a portion of biomass carbon into more recalcitrant forms, but the degree of permanence varies with feedstock composition, pyrolysis conditions, and the environment in which the biochar is stored [113]. Recent stability assessment studies emphasize that permanence should be evaluated using measurable chemical and physical indicators rather than assumed to be an intrinsic quality of biochar [17,22]. A comparison of multiple stability assessment methods for biochars produced from soybean straw, wood sawdust, and *C. vulgaris* across 300–800 °C found that pyrolysis temperature was a primary factor controlling stability, with most indicators significantly related to temperature [85]. However, the authors also showed that elemental ratios, proximate analysis, X-ray photoelectron spectroscopy, solid-state ^{13}C nuclear magnetic resonance, thermogravimetric methods, and chemical oxidation approaches can produce different stability estimates, highlighting the need to use multiple indicators when evaluating sequestration potential [85].

Among the most common practical proxies for long-term carbon persistence are elemental ratios. The H/C and O/C ratios decline as pyrolysis temperature increases because dehydration, decarboxylation, and devolatilization remove hydrogen and oxygen-rich functional groups, leaving a more carbon-dense material [99,114]. Biochars with low H/C and O/C ratios generally exhibit greater aromaticity and resistance to microbial degradation, allowing them to persist over the long term in soil applications. H/C ratios below 0.7 are commonly used to distinguish sufficiently carbonized biochar from raw biomass, while values below 0.4 indicate greater stability and sequestration potential. O/C ratios below 0.2 have been associated with half-lives exceeding 1000 years, while O/C ratios between 0.2 and 0.6 correspond to intermediate persistence on the order of 100 to 1000 years [17,61,82]. Applying these thresholds to algal biochar, however, requires care given that they were derived largely from low-ash lignocellulosic biochars [114]. The carbonate minerals that line algal biochar pores add inorganic carbon that inflates stability estimates when ratios are based on total rather than organic carbon [89,91]. An organic-carbon-based H/C ratio paired with an independent method such as proximate analysis or random reflectance would give a more defensible estimate of permanence for algal biochar [85].

Permanence also depends on where the biochar is stored. Soil application is the most common endpoint for biochar, but it is not the only option, and its permanence is not guaranteed. Adding biochar to soil can change the decomposition rate of native organic matter, a process known as priming [25,26]. Biochar can suppress this decomposition over time, but it can also accelerate it. Algal biochar may be more prone to accelerating it, given its higher nitrogen and phosphorus contents, which stimulate microbial activity [25,26,89,95]. We found no study that has measured priming by algal biochar directly, so it remains a key uncertainty. A meta-analysis of mostly terrestrial biochars found a small overall negative priming effect (−3.8%) but strong positive priming (+20.8%) in sandy soils, indicating that soil type can reverse the direction of the effect [86]. Biochar can also be stored in construction materials like concrete, asphalt, and other manufactured products. This is a promising pathway because construction materials are produced at large scales and embed biochar in a mineral matrix that helps protect it from degradation [114,115].

Several further factors determine whether algal biochar can act as a durable and safe soil carbon store. First, most persistence estimates come from laboratory incubations and model extrapolation rather than field measurement. A meta-analysis of isotope-labelled studies estimated mean residence times of about 108 days for a small labile pool (about 3% of biochar carbon) and about 556 years for the remaining 97% [86], and the IPCC methodology assumes that 65–89% of biochar organic carbon remains after 100 years, depending on production temperature [116]. Neither estimate is based on algal biochars. To our knowledge, the long-term carbon stability

of algal biochar has not yet been directly established through isotope-labelled incubations or field trials, so stability values derived for terrestrial biochars should be regarded as provisional when applied to algal feedstocks. Second, effects on nitrous oxide (N_2O), a far more potent greenhouse gas than CO_2, belong in the carbon balance. Meta-analyses report average reductions in soil N_2O emissions of 54% across laboratory and field studies [117] and 38% in a later, larger analysis, in which reductions tended to become negligible after one year [118]. Whether nitrogen-rich algal biochars behave similarly is unknown, and their labile nitrogen could plausibly supply substrate for nitrification and denitrification. Third, pyrolysis can form polycyclic aromatic hydrocarbons (PAHs), and wastewater-grown algae can concentrate heavy metals (Section 6). Both are regulated. Under the EU Fertilizing Products Regulation, pyrolysis and gasification materials may contain no more than 6 mg kg^{-1} dry matter of the 16 priority PAHs (PAH16) [119], and organic soil improvers are limited to, for example, 2 mg kg^{-1} cadmium, 1 mg kg^{-1} mercury, 120 mg kg^{-1} lead and 40 mg kg^{-1} inorganic arsenic [120]. Carbon crediting protocols likewise require PAH and heavy metal safeguards for soil application [121]. Algal biochars intended for soil should therefore be tested against these thresholds, and those that fail them may be better directed to non-soil storage.

5. Environmental and Economic Analysis

Although algae-to-biochar-based carbon sequestration systems have strong theoretical potential, large-scale deployment ultimately depends on whether they deliver measurable environmental benefits while remaining economically viable. Effective CDR itself does not necessarily guarantee a net climate benefit, because emissions generated throughout the system's life cycle can offset a significant fraction of the carbon captured. Therefore, evaluating these systems requires a systems-level perspective that accounts for environmental impacts and economic performance across the entire process boundary.

5.1. Life Cycle Assessment (LCA)

For an algal system to be net-negative, the carbon captured during growth must exceed the carbon emitted across the full process chain. LCA provides the standard tool for evaluating this balance, accounting for all material inputs, energy use, and emissions from raw material sourcing through end-of-life disposal [122,123]. In the context of algal-biochar systems, a robust LCA considers the overall carbon balance of emissions generated throughout cultivation, biomass processing, and downstream conversion, in addition to upstream processes (sourcing, construction, etc.) and the system's disposal at the end of life.

Comparing LCA results across algal biochar studies is difficult because studies define their systems differently. Functional units range from a kilogram of biochar to a megajoule of biofuel. Many algal LCAs evaluate biofuels, which re-release the fixed carbon when the fuel is burned, so their climate benefit comes from displacing fossil fuels rather than durable storage [124]. Results from these studies cannot be compared directly with those for biochar, even when both report net-negative values. System boundaries also vary. Most algal LCAs stop at biomass production or fuel conversion and only a few extend through pyrolysis to biochar storage [80,124–129]. Where biochar is included, the boundary choice can change the sign of the result. In one seaweed (*Saccharina latissima*) biorefinery that pyrolyzed its solid residues to biochar, cradle-to-gate impacts were net negative at −3.71 to −2.17 kg CO_2-eq kg^{-1} of biopolymer product but rose to 2.99–5.40 kg CO_2-eq kg^{-1} on a cradle-to-grave basis when the biopolymer was assumed to be landfilled at end of life [130]. Methodological and process choices add further spread. After harmonizing functional unit, scope and allocation, 37 published biochar life cycle inventories gave cradle-to-gate removals of 0.20–1.51 kg CO_2-eq per kg of feedstock (5th–95th percentile), and a Monte Carlo analysis showed that a poorly configured system is more likely than not to be a net emitter, with the emission intensity of energy inputs and the fate of the bio-oil accounting for nearly all of the difference between best- and poor-practice scenarios [131]. Purpose-grown feedstocks in that comparison averaged about 0.3 kg CO_2-eq kg^{-1} of removal compared with 0.75 kg CO_2-eq kg^{-1} for residues, because of the energy and fertilizer used to grow them [131], a pattern directly relevant to algae cultivated for carbon removal.

Only a few assessments follow algal biomass all the way to biochar storage. Besides the seaweed biorefinery above [130], an assessment of an air-capture algae coating with pyrolysis to biochar found that the system emitted 0.51–0.73 kg CO_2-eq per kg of CO_2 captured, leaving net removal efficiencies of 27–49% [45]. To show how the main terms combine, Table 6 sets out an illustrative order-of-magnitude carbon balance for one tonne of dry microalgal biomass grown on atmospheric or biogenic CO_2. Growth fixes about 1.8 t CO_2 [132]. For stable biochars, the fraction of feedstock carbon that is both retained in the char and still present after 100 years has been estimated at 25–50% for plant-based feedstocks [22]. No equivalent estimate yet exists for algal biochar; applying the plant-based range, roughly 0.46–0.92 t CO_2 would remain stored. Drying alone can consume a large part of this: if the biomass is mechanically dewatered to 20% solids, about 4 t of water must be evaporated, requiring at

least about 9 GJ of heat, which would emit about 0.5 t CO_2 if supplied by natural gas [133]. Before the energy for cultivation, harvesting, pyrolysis and transport is counted, the balance therefore ranges from a net removal of about 0.4 t CO_2 to a small net emission. Table 6 is not, however, a complete carbon balance. The excluded terms (energy for cultivation, mixing, harvesting, pyrolysis and transport; embodied emissions of infrastructure and nutrients; and any soil N_2O or priming effects) would all add to the emission side, so the values in Table 6 represent an upper bound on net removal rather than an estimate of it. Whether a specific algae-to-biochar system is net-negative can only be established through a full life cycle assessment of that system. This simple calculation explains why the LCAs above diverge so widely and why waste heat, low-carbon energy and wet conversion routes (Section 3.3) are necessary, but not by themselves sufficient, conditions for a net-negative outcome.

Table 6. Illustrative carbon balance for 1 t of dry microalgal biomass (100-year horizon; order-of-magnitude values). Negative values denote net removal. Energy for cultivation, harvesting, pyrolysis and transport is excluded. The net balance is therefore a best case (an upper bound on removal), not a full life-cycle estimate.

Term	Value	Basis
CO_2 fixed during growth	≈1.8 t CO_2	Biomass ≈ 50% C by dry weight [132]
Feedstock C stored after 100 years (char C yield × persistence)	25–50%	Stable plant-based biochars; not yet established for algal biochar [22]
CO_2 stored in biochar after 100 years	≈0.46–0.92 t CO_2	Row 1 × Row 2
Water evaporated (paste dewatered to 20% solids)	≈4 t	Illustrative assumption
Minimum heat for evaporation	≈9 GJ	Latent heat of water ≈ 2.26 GJ t^{-1}
CO_2 if heat is supplied by natural gas	≈0.5 t CO_2	56.1 t CO_2 TJ^{-1} [133]
Net balance (drying only; upper bound on removal)	≈−0.4 to +0.05 t CO_2	Row 6 − Row 3

Despite these differences, consistent patterns emerge in which stages drive environmental impact. Because cultivation, harvesting, and dewatering are common to both biofuel and biochar pathways, LCAs of algal biofuels remain informative for these upstream stages. Recent algal LCAs consistently show that environmental impacts are not evenly distributed across the process chain but are concentrated at the most energy-intensive stages [124]. In large-scale marine microalgae systems in India, cultivation and harvesting alone dominated environmental impact across multiple categories including global warming potential, eutrophication, and ecotoxicity, largely due to the energy demand associated with pumping, aeration, and biomass separation [125]. The energy requirements for cultivation, harvesting, and drying are particularly significant drivers of the total environmental burden [124]. The dilute nature of algal cultures means that harvesting and dewatering remain energy-intensive because the biomass requires substantial processing before downstream thermochemical conversion can occur [132]. Additional burdens arise from artificial lighting, pumping, nutrient production, and reactor operation, especially in closed cultivation systems that rely on fossil-fuel-based energy units [124–127].

One of the clearest paths to improving these outcomes is integration. Rather than treating cultivation, carbon capture, and downstream conversion as independent processes, integrated approaches treat the system as a whole, recovering waste heat, recycling nutrients, and maximizing carbon transfer efficiency across process boundaries [108]. Waste products from industrial and commercial activities can be integrated into algal systems, reframing disposal liabilities as functional inputs and reducing upstream burdens [124]. Wastewater-based cultivation is the most studied example. Using municipal or industrial wastewater as the nutrient source has been shown to produce significantly lower environmental impacts across multiple indicators, including acidification and eutrophication potential, and TEA studies suggest it reduces operating costs by up to 52% compared to freshwater systems [108,124,128]. Pairing wastewater cultivation with industrial flue gas as the CO_2 source further improves outcomes, as the system now substitutes two conventional inputs with waste streams simultaneously. Recent algal biofuel LCAs using this approach have reported greenhouse gas emissions as low as approximately −4 g CO_2-eq MJ^{-1} when the supplied carbon source is biogenic, highlighting the potential of true net-negative fuel production under the right conditions [18,129]. These values cannot be directly translated to algae-to-biochar systems as they reflect avoided fossil fuel emissions rather than durable storage, although the upstream savings from waste stream integration can apply to both. However, relying on two waste streams also introduces new constraints. Wastewater composition varies over daily and seasonal cycles and often requires pretreatment before it can support stable algal growth [16,108]. Furthermore, wastewater can also contain heavy metals that can be carried into the biochar if not addressed correctly [14]. Flue gas must likewise be cooled and cleaned of SO_x and NO_x, and its supply also fluctuates depending on plant operation [18].

Energy sourcing may be the largest single variable in the life-cycle performance of an algal system. The same cultivation setup can produce very different outcomes depending on what electricity mix powers it. Onsite solar or low-carbon grid power dramatically reduces the operational footprint compared to fossil-fuel-based electricity [80,134]. Heat and mass integration have been shown to reduce GHG emissions by roughly 50% and lower costs by about 18% relative to non-integrated baselines [44]. The strongest LCA outcomes, then, come from systems that are designed around local conditions: wastewater availability, carbon source, climate, and electricity mix are treated as site-specific inputs rather than fixed parameters.

5.2. Policy and Carbon Markets

Beyond technological development, economic and regulatory frameworks will likely determine whether algae-based carbon sequestration systems achieve large-scale deployment. Current markets increasingly recognize the value of carbon removal pathways; however, implementation of biologically based carbon sequestration remains limited due to challenges associated with measurement, verification, and permanence [135]. Biologically derived storage systems often involve greater uncertainty than their engineered geological counterparts [135]. To make algal storage systems viable, standardized, cost-effective carbon accounting frameworks are essential. Certification frameworks that treat temporary carbon removal and long-term sequestration equally undermine the incentive structures needed to scale emerging carbon removal pathways during their early development and commercialization. Rather, successful frameworks must establish monitoring, reporting, and verification (MRV) protocols and propose standardized approaches to evaluate permanence, additionality, and leakage [136]. Carbon markets are increasingly incorporating these factors into project valuation, with high-permanence pathways typically receiving greater market value because they reduce future uncertainty.

Biochar-based carbon removal projects are gaining traction due to their potential to persist for centuries to millennia under optimal conditions [26]. Several methodologies now credit biochar-based removal, including Verra's VM0044, Puro.earth, and Isometric, with each estimating permanence primarily from the H/C_{org} ratio [102,121,137]. Their durability assumptions differ, ranging from 100 years of storage in soil under VM0044 to as long as 1000 years under Isometric when reflectance data demonstrate highly stable carbon [121]. For algal systems, feedstock eligibility differs more sharply between methodologies. VM0044 credits only waste biomass and its proposed revision extends eligibility only to feedstocks that are not purpose-grown, so algae cultivated for carbon removal would not qualify [102]. Isometric applies separate eligibility criteria to purpose-grown feedstocks through its Biomass Feedstock Accounting Module [121], while Puro.earth's 2025 Biochar Methodology classifies algae under a dedicated aquatic biomass category [137], and a facility using purpose-grown microalgae has passed Puro.earth's preliminary eligibility assessment [138]. More broadly, no current methodology is designed specifically for algae-to-biochar systems. Future policy development will need to account for the energy demands of cultivation and dewatering, distinguish atmospheric from fossil-derived CO_2 inputs, and apply stability metrics suited to high-ash algal biochar to credibly quantify and incentivize algal carbon removal. Genetically engineered strains raise a further gap. None of the methodologies reviewed here addresses genetically engineered (GE) organisms explicitly, yet outdoor cultivation of GE algae already requires regulatory review, such as EPA oversight in the United States, together with secondary containment measures [64,65]. Credits are also contingent on documented environmental and social safeguards [121], so an unaddressed risk of GE strain release could delay validation or exclude a project. Until crediting frameworks specify how the use, containment and monitoring of GE strains are to be documented, projects built around engineered strains will carry additional regulatory uncertainty and may in practice be limited to closed photobioreactors.

6. Challenges and Future Prospects

While algae-to-biochar systems have significant potential as net-negative carbon strategies, several technical and economic barriers continue to limit large-scale deployment despite advances in cultivation systems, strain engineering, and thermochemical conversion.

6.1. Drying Requirements

Due to the inherently high moisture content of microalgal biomass, drying and dewatering remain significant challenges for large-scale deployment [132]. These steps are among the most energy-intensive stages in the process chain and can substantially increase operational costs and life-cycle emissions when conventional drying methods are used [124]. Since downstream processes, such as pyrolysis, typically require lower moisture content for efficient conversion, optimizing biomass processing methods will be critical [124,132]. Integration of waste heat recovery, low-energy harvesting techniques, and alternative wet processing pathways may help reduce these

environmental and economic burdens. As Table 6 shows, evaporating water from a mechanically dewatered paste with fossil heat could offset most or all of the carbon stored. Wet conversion by HTC avoids thermal drying but yields less stable hydrochar (Section 3.3), so the choice between drying and wet conversion is itself a permanence trade-off.

6.2. Heavy Metal Accumulation

Although wastewater-integrated algal systems provide environmental benefits through nutrient recovery and contaminant removal, heavy metal accumulation remains a significant challenge. Microalgae can effectively assimilate metals such as cadmium (Cd), lead (Pb), chromium (Cr), and arsenic (As), which may subsequently concentrate within harvested biomass and persist through downstream conversion processes [14]. During pyrolysis, these contaminants can become retained within the biochar matrix and potentially affect its environmental safety and agricultural applicability [139]. Therefore, improved feedstock monitoring and standardized quality assessment protocols will be important for ensuring long-term sustainability.

Mitigation options exist at each stage of the chain. Before cultivation, wastewater sources can be screened and pretreated, and metal concentrations in the harvested biomass monitored against the limits that apply to the intended end use (Section 4.2). During conversion, pyrolysis conditions can be tuned: pyrolysis tends to transform heavy metals from bioavailable into more stable oxidizable and residual forms, but cadmium can volatilize into the gas and oil phases above about 600 °C, so emission control is needed [140]. Co-pyrolysis with clean lignocellulosic biomass can further dilute and immobilize metals and has reduced the environmental risk of sewage-sludge biochars by up to 93% [140]. After production, biochars that exceed soil-application limits can be directed to non-soil uses such as construction materials [115], subject to the leaching requirements of those applications. These measures were developed mainly for sewage sludge, and their effectiveness for algal biomass still needs to be confirmed.

6.3. Scale-Up

Scale-up remains a major challenge for the commercial deployment of algae-based carbon sequestration systems because favorable laboratory performance does not always translate to large-scale operation. While controlled studies frequently report high biomass productivities and carbon fixation rates, outdoor and commercial systems often exhibit reduced performance due to environmental fluctuations, contamination risks, nutrient variability, and increased operational complexity [80]. Additionally, large-scale deployment requires significant capital investment, especially given the novelty of the process. Future research should prioritize demonstration-scale studies together with region-specific optimization strategies to improve scalability and real-world performance.

6.4. Strain Stability Outdoors

Traits that perform well in the laboratory must persist in outdoor cultures exposed to competitors, grazers and fluctuating conditions. The only EPA-sanctioned outdoor trial of an engineered alga maintained its introduced traits for 50 days, which is short compared with the multi-year operation of a carbon removal facility, and engineered cells dispersed beyond the cultivation ponds [64]. Long-duration outdoor trials that track both trait stability and culture composition are needed before engineered strains can be relied on for carbon removal.

6.5. Reactor Compatibility with Wet Algal Feedstock

Most commercial pyrolysis reactors are designed for relatively dry, coarse lignocellulosic feedstocks. Dewatered algal biomass is instead a fine, high-moisture paste with high ash and, for marine species, high salt content, which can complicate feeding, promote fouling and lower char quality. Chlorine is also regulated: EU fertilising products containing pyrolysis materials may contain no more than 30 g kg^{-1} dry matter of chloride [119]. Feeding systems, reactor designs and dryer integration will need to be adapted to algal feedstocks and validated at pilot scale (Section 3.1).

6.6. Supply Chain Logistics

Algal biomass is harvested as a dilute suspension that is perishable and costly to store or transport wet, so harvesting, dewatering and conversion generally need to be located close to cultivation. This differs from terrestrial biochar supply chains, in which dry residues can be stockpiled and moved. Offshore seaweed farming adds the cost and emissions of marine operations and transport to shore, which vary strongly with location [141]. Facility

siting must therefore weigh not only light, temperature and water, but also proximity to CO_2 sources, waste heat, pyrolysis capacity and biochar end users.

6.7. Resource Requirements at Scale

The 7–9 Gt CO_2 $year^{-1}$ of removal cited in Section 1 refers to all CDR methods combined, but an order-of-magnitude estimate shows what even part of this would demand from algae-to-biochar systems. Using Table 6, storing 1 Gt CO_2 for 100 years would require roughly 1.1–2.2 Gt of dry algal biomass per year, before process emissions are subtracted. At an annual average productivity of 20 g m^{-2} d^{-1}, which matches the peak values reported with CO_2 supplementation and is therefore optimistic [40,41], about 73 t ha^{-1} $year^{-1}$ of biomass would be produced, implying roughly 15–30 million ha of cultivation per Gt CO_2 stored; lower, more typical productivities would raise this proportionally. Nutrient demand is also large. Based on the approximate molecular formula of microalgal biomass ($CO_{0.48}H_{1.83}N_{0.11}P_{0.01}$), biomass contains about 6.6% nitrogen and 1.3% phosphorus by dry weight [132], so 1.1–2.2 Gt of biomass would contain about 70–145 Mt of nitrogen, comparable to the roughly 109 Mt of nitrogen fertilizer used in global agriculture in 2021 [142], unless nutrients come from wastewater or are recycled from process streams. Water demand depends strongly on system type, with evaporative losses dominating in open ponds (Table 1) and strong regional variation in water stress [80]. Finally, drying a 20%-solids paste would require on the order of 10–20 EJ $year^{-1}$ of heat per Gt CO_2 stored (Table 6) unless waste heat or wet conversion is used. These figures rest on simplifying assumptions: (i) the storage range in Table 6, that is, 25–50% of feedstock carbon remaining after 100 years, which is derived from plant-based biochars; (ii) a biomass composition represented by the approximate microalgal formula (about 50% carbon, 6.6% nitrogen and 1.3% phosphorus); (iii) a uniform year-round productivity of 20 g m^{-2} d^{-1}; (iv) no recycling of nutrients or water; and (v) drying from 20% solids at the theoretical minimum heat demand, without dryer losses. Process emissions are excluded, and including them would increase the biomass, land and nutrient required per tonne of CO_2 stored. These figures are intended to indicate orders of magnitude rather than to predict deployment. These estimates suggest that algae-to-biochar systems are more plausibly a regional contributor, built around waste nutrients, waste heat and biogenic CO_2, than a gigatonne-scale pathway on their own. The model-estimated share of revenue from co-products (Section 4.1) also depends on markets that are small compared with this scale of biomass production.

6.8. Research Priorities

The challenges above translate into a set of research needs that differ in scale and in who must act on them. Table 7 groups them into laboratory research, pilot demonstration, and policy and methodology development, and links each to the section in which the underlying gap is discussed.

Table 7. Prioritized research needs for algae-to-biochar carbon removal.

Stage	Research Priority	Sections
Laboratory research	Measure carbon yield and organic-carbon-based H/C of microalgal and macroalgal biochars across pyrolysis temperatures, with an independent stability method	3.1, 4.2
Laboratory research	Test priming, N_2O emissions and PAH and heavy-metal behavior of algal biochar in soil incubations	4.2
Laboratory research	Select and engineer strains for ash, nitrogen and cell wall traits relevant to char rather than for lipid content alone	2.1
Pilot demonstration	Operate integrated cultivation–dewatering–pyrolysis (or HTC) pilots, including hybrid PBR–pond systems, with measured mass and energy balances	2.3, 3.1, 3.3
Pilot demonstration	Conduct multi-criteria LCA and TEA of pilot-scale chains with boundaries that extend to biochar storage	5.1
Pilot demonstration	Run multi-year outdoor trials of trait stability and containment for engineered strains; measure field persistence of algal biochar	2.1, 4.2, 6
Policy and methodology	Develop algae-specific MRV protocols that attribute CO_2 sources (atmospheric, biogenic, fossil) and use stability metrics suited to high-ash chars	2, 5.2
Policy and methodology	Clarify eligibility of purpose-grown and wastewater-grown algae and of engineered strains in crediting methodologies	5.2
Policy and methodology	Define contaminant testing for algal biochar in soil and non-soil uses	4.2, 6

7. Conclusions

The growing need for durable carbon removal solutions highlights the importance of developing scalable NETs beyond conventional mitigation strategies. This review evaluated the potential of optimized algal systems integrated with biochar production as a pathway for long-term carbon sequestration. Compared to traditional biological systems, algae offer several advantages, including high photosynthetic efficiency, rapid biomass productivity, and the ability to utilize non-arable land, wastewater resources, and alternative carbon streams. These advantages are qualified, however: cultivation on non-arable land or in coastal waters still occupies space with its own biodiversity and ecosystem value, and open systems lose freshwater through evaporation, so land and water trade-offs are reduced rather than eliminated. Optimized algal systems coupled with biochar production could provide a pathway for durable carbon removal, but their net-negative status is conditional rather than inherent and can only be confirmed by full life cycle assessments of operating systems. Unlike algal biofuels, which re-release the fixed carbon on combustion, this approach could store part of it for centuries while providing soil benefits, although such long-term stability has yet to be directly demonstrated for algal biochar. Several bottlenecks must be resolved first. Technically, drying and dewatering can offset much or all of the stored carbon when fossil heat is used; most performance data come from laboratory-scale cultivation and pyrolysis; algal biochars are high in ash and nitrogen, which complicates stability assessment and raises questions about NO_x-precursor emissions, priming and N_2O; and wastewater-grown biomass can carry heavy metals into the char. Economically, closed photobioreactors remain costly, and favorable cascading biorefinery economics rest on modeled co-product markets. From a regulatory perspective, current crediting methodologies were not designed for algal feedstocks, may exclude purpose-grown biomass and do not address engineered strains, and only CO_2 of atmospheric or sustainably sourced biogenic origin can support a removal claim. While additional technological and economic development is still required, these systems may play a role within future carbon removal portfolios, most plausibly at regional scales where waste nutrients, waste heat and biogenic CO_2 are available. To realize this potential, future work should move from general goals to specific tests: multi-criteria LCAs of pilot-scale algae-to-biochar chains with boundaries that extend to biochar storage; experimental measurement of priming, N_2O emissions and field persistence for algal biochar; algae-specific MRV protocols that attribute CO_2 sources and use stability metrics suited to high-ash chars; and pilot facilities that couple hybrid cultivation with pyrolysis or hydrothermal conversion under measured mass and energy balances (Table 7).

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References

1. Murphy, R. What is undermining climate change mitigation? How fossil-fuelled practices challenge low-carbon transitions. *Energy Res. Soc. Sci.* **2024**, *108*, 103390. https://doi.org/10.1016/j.erss.2023.103390.
2. Yoro, K.O.; Daramola, M.O. CO_2 emission sources, greenhouse gases, and the global warming effect. In *Advances in Carbon Capture*; Woodhead Publishing: Cambridge, UK, 2020; pp. 3–28. https://doi.org/10.1016/b978-0-12-819657-1.00001-3.
3. International Energy Agency. *Global Energy Review 2025*; IEA: Paris, France, 2025. Available online: https://www.iea.org/reports/global-energy-review-2025 (accessed on 19 October 2026).
4. National Oceanic and Atmospheric Administration. Carbon Cycle Greenhouse Gases: Trends in CO_2, CH_4, N_2O, SF_6. 2025. Available online: https://gml.noaa.gov/ccgg/trends/mlo.html (accessed on 19 October 2026).
5. Matthews, H.D.; Wynes, S. Current global efforts are insufficient to limit warming to 1.5 °C. *Science* **2022**, *376*, 1404–1409. https://doi.org/10.1126/science.abo3378.
6. Hoegh-Guldberg, O.; Jacob, D.; Taylor, M.; et al. The human imperative of stabilizing global climate change at 1.5 °C. *Science* **2019**, *365*, eaaw6974. https://doi.org/10.1126/science.aaw6974.

7. Smith, S.M.; Geden, O.; Gidden, M.J.; et al. *The State of Carbon Dioxide Removal*, 2nd ed.; OSF: Charlottesville, VA, USA, 2024. https://doi.org/10.17605/osf.io/f85qj.
8. Bhatia, M.; Gugnani, R.; Yaqub, M.Z.; et al. Emission reduction strategies and negative emission solutions-pathways, drivers, and challenges. *J. Clean. Prod.* **2025**, *500*, 145263. https://doi.org/10.1016/j.jclepro.2025.145263.
9. Xu, Y.; Huang, G.; Liu, Y.; et al. Exploring the role of negative emission technologies in regional power system planning toward carbon net zero—A case study for the province of Saskatchewan, Canada. *Energy* **2025**, *326*, 136351. https://doi.org/10.1016/j.energy.2025.136351.
10. Zhang, Y.; Jackson, C.; Krevor, S. The feasibility of reaching gigatonne scale CO_2 storage by mid-century. *Nat. Commun.* **2024**, *15*, 6913. https://doi.org/10.1038/s41467-024-51226-8.
11. Borchers, M.; Förster, J.; Thrän, D.; et al. A Comprehensive Assessment of Carbon Dioxide Removal Options for Germany. *Earth's Future* **2024**, *12*, e2023EF003986. https://doi.org/10.1029/2023ef003986.
12. Smith, P.; Adams, J.; Beerling, D.J.; et al. Land-Management Options for Greenhouse Gas Removal and Their Impacts on Ecosystem Services and the Sustainable Development Goals. *Annu. Rev. Environ. Resour.* **2019**, *44*, 255–286. https://doi.org/10.1146/annurev-environ-101718-033129.
13. Ashour, M.; Mansour, A.T.; Alkhamis, Y.A.; et al. Usage of Chlorella and diverse microalgae for CO_2 capture—Towards a bioenergy revolution. *Front. Bioeng. Biotechnol.* **2024**, *12*, 1387519. https://doi.org/10.3389/fbioe.2024.1387519.
14. Leong, Y.K.; Chang, J.S. Microalgae-based biochar production and applications: A comprehensive review. *Bioresour. Technol.* **2023**, *389*, 129782. https://doi.org/10.1016/j.biortech.2023.129782.
15. Gao, G.; Beardall, J.; Jin, P.; et al. A review of existing and potential blue carbon contributions to climate change mitigation in the Anthropocene. *J. Appl. Ecol.* **2022**, *59*, 1686–1699. https://doi.org/10.1111/1365-2664.14173.
16. Nguyen, L.N.; Aditya, L.; Vu, H.P.; et al. Nutrient Removal by Algae-Based Wastewater Treatment. *Curr. Pollut. Rep.* **2022**, *8*, 369–383. https://doi.org/10.1007/s40726-022-00230-x.
17. Li, S.; Tasnady, D. Biochar for Soil Carbon Sequestration: Current Knowledge, Mechanisms, and Future Perspectives. *C* **2023**, *9*, 67. https://doi.org/10.3390/c9030067.
18. Almomani, F.; Judd, S.; Bhosale, R.R.; et al. Intergraded wastewater treatment and carbon bio-fixation from flue gases using Spirulina platensis and mixed algal culture. *Process Saf. Environ. Prot.* **2019**, *124*, 240–250. https://doi.org/10.1016/j.psep.2019.02.009.
19. Arora, N.; Tripathi, S.; Bhatnagar, P.; et al. Algal-based biochar and hydrochar: A holistic and sustainable approach to wastewater treatment. *Chem. Eng. J.* **2024**, *496*, 153953. https://doi.org/10.1016/j.cej.2024.153953.
20. Srimongkol, P.; Sangtanoo, P.; Songserm, P.; et al. Microalgae-based wastewater treatment for developing economic and environmental sustainability: Current status and future prospects. *Front. Bioeng. Biotechnol.* **2022**, *10*, 904046. https://doi.org/10.3389/fbioe.2022.904046. Correction in *Front. Bioeng. Biotechnol.* **2022**, *10*, 1048819.
21. Chiaramonti, D.; Lehmann, J.; Berruti, F.; et al. Biochar is a long-lived form of carbon removal, making evidence-based CDR projects possible. *Biochar* **2024**, *6*, 81. https://doi.org/10.1007/s42773-024-00366-7.
22. Rodrigues, L.; Budai, A.; Elsgaard, L.; et al. The importance of biochar quality and pyrolysis yield for soil carbon sequestration in practice. *Eur. J. Soil Sci.* **2023**, *74*, e13396. https://doi.org/10.1111/ejss.13396.
23. Brunner, C.; Hausfather, Z.; Knutti, R. Durability of carbon dioxide removal is critical for Paris climate goals. *Commun. Earth Environ.* **2024**, *5*, 645. https://doi.org/10.1038/s43247-024-01808-7. Correction in *Commun. Earth Environ.* **2024**, *5*, 713.
24. Dynarski, K.A.; Bossio, D.A.; Scow, K.M. Dynamic Stability of Soil Carbon: Reassessing the "Permanence" of Soil Carbon Sequestration. *Front. Environ. Sci.* **2020**, *8*, 514701. https://doi.org/10.3389/fenvs.2020.514701.
25. Han, W.; Lai, Y.; Ji, H. Biochar-driven soil carbon sequestration: Priming effects and emission reduction. *Environ. Sci. Process. Impacts* **2025**, *27*, 3725–3745. https://doi.org/10.1039/d5em00500k.
26. Joseph, S.; Cowie, A.L.; Van Zwieten, L.; et al. How biochar works, and when it doesn't: A review of mechanisms controlling soil and plant responses to biochar. *GCB Bioenergy* **2021**, *13*, 1731–1764. https://doi.org/10.1111/gcbb.12885.
27. Tan, X.; Zhang, F.; Wang, H.; et al. The magic of algae-based biochar: Advantages, preparation, and applications. *Bioengineered* **2023**, *14*, 2252157. https://doi.org/10.1080/21655979.2023.2252157.
28. Mustapha, S.I.; Isa, Y.M. Thermal Conversion of Microalgae into Biochar: A Review on Processes, Properties, and Applications. *ChemBioEng Rev.* **2025**, *12*, e202400035. https://doi.org/10.1002/cben.202400035.
29. Derwenskus, F.; Metz, F.; Gille, A.; et al. Pressurized extraction of unsaturated fatty acids and carotenoids from wet *Chlorella vulgaris* and *Phaeodactylum tricornutum* biomass using subcritical liquids. *GCB Bioenergy* **2019**, *11*, 335–344. https://doi.org/10.1111/gcbb.12563.
30. Bligh, M.; Nguyen, N.; Buck-Wiese, H.; et al. Structures and functions of algal glycans shape their capacity to sequester carbon in the ocean. *Curr. Opin. Chem. Biol.* **2022**, *71*, 102204. https://doi.org/10.1016/j.cbpa.2022.102204.
31. Buck-Wiese, H.; Andskog, M.A.; Nguyen, N.P.; et al. Fucoid brown algae inject fucoidan carbon into the ocean. *Proc. Natl. Acad. Sci. USA* **2023**, *120*, e2210561119. https://doi.org/10.1073/pnas.2210561119.

32. Zhao, X.; Wang, X.; Zhang, J.; et al. Sustainable production of ulvan from green macroalgae and its potential in carbon sequestration: A critical review. *Int. J. Biol. Macromol.* **2025**, *330*, 147978. https://doi.org/10.1016/j.ijbiomac.2025.147978.
33. Pari, R.F.; Uju, U.; Hardiningtyas, S.D.; et al. Ulva Seaweed-Derived Ulvan: A Promising Marine Polysaccharide as a Sustainable Resource for Biomaterial Design. *Mar. Drugs* **2025**, *23*, 56. https://doi.org/10.3390/md23020056.
34. Kim, K.H.; Parrow, M.W.; Kheirkhah Sangdeh, P. Microalgae-integrated building enclosures: A nature-based solution for carbon sequestration. *Front. Built Environ.* **2025**, *11*, 1574582. https://doi.org/10.3389/fbuil.2025.1574582.
35. Perin, G.; Gambaro, F.; Morosinotto, T. Knowledge of Regulation of Photosynthesis in Outdoor Microalgae Cultures Is Essential for the Optimization of Biomass Productivity. *Front. Plant Sci.* **2022**, *13*, 846496. https://doi.org/10.3389/fpls.2022.846496.
36. Singh Chauhan, D.; Sahoo, L.; Mohanty, K. Acclimation-driven microalgal cultivation improved temperature and light stress tolerance, CO_2 sequestration and metabolite regulation for bioenergy production. *Bioresour. Technol.* **2023**, *385*, 129386. https://doi.org/10.1016/j.biortech.2023.129386.
37. Maltsev, Y.; Maltseva, K.; Kulikovskiy, M.; et al. Influence of Light Conditions on Microalgae Growth and Content of Lipids, Carotenoids, and Fatty Acid Composition. *Biology* **2021**, *10*, 1060. https://doi.org/10.3390/biology10101060.
38. Weise, T.; Grewe, C.; Pfaff, M. Experimental and Model-Based Analysis to Optimize Microalgal Biomass Productivity in a Pilot-Scale Tubular Photobioreactor. *Front. Bioeng. Biotechnol.* **2020**, *8*, 453. https://doi.org/10.3389/fbioe.2020.00453.
39. Li, C.T.; Eng, R.; Zuniga, C.; et al. Optimization of nutrient utilization efficiency and productivity for algal cultures under light and dark cycles using genome-scale model process control. *npj Syst. Biol. Appl.* **2023**, *9*, 7. https://doi.org/10.1038/s41540-022-00260-7.
40. Chunzhuk, E.A.; Grigorenko, A.V.; Kiseleva, S.V.; et al. The Influence of Elevated CO_2 Concentrations on the Growth of Various Microalgae Strains. *Plants* **2023**, *12*, 2470. https://doi.org/10.3390/plants12132470.
41. Palikrousis, T.L.; Kalamaras, S.D.; Samaras, P. Effect of Carbon Dioxide on the Growth and Nutrient Uptake of the Microalgae *Chlorella sorokiniana* from Digestate. *Water* **2025**, *17*, 2674. https://doi.org/10.3390/w17182674.
42. Azhand, N.; Sadeghizadeh, A.; Rahimi, R. Effect of superficial gas velocity on CO_2 capture from air by Chlorella vulgaris microalgae in an Airlift photobioreactor with external sparger. *J. Environ. Chem. Eng.* **2020**, *8*, 104022. https://doi.org/10.1016/j.jece.2020.104022.
43. Dayanath, W.M.S.; Ilankoon, I.M.S.K.; Syed Ali, S.A.; et al. Microalgae-based direct air capture systems for carbon dioxide emission reductions in developing megacities: A techno-economic and policy analysis. *J. Environ. Chem. Eng.* **2026**, *14*, 122763. https://doi.org/10.1016/j.jece.2026.122763.
44. D'Souza, S.; Johnston, J.; Thomas, V.M.; et al. Integrating direct air capture with algal biofuel production to reduce cost, energy, and GHG emissions. *J. CO_2 Util.* **2024**, *86*, 102911. https://doi.org/10.1016/j.jcou.2024.102911.
45. Cole, G.M.; Greene, J.M.; Quinn, J.C.; et al. Integrated techno-economic and life cycle assessment of a novel algae-based coating for direct air carbon capture and sequestration. *J. CO_2 Util.* **2023**, *69*, 102421. https://doi.org/10.1016/j.jcou.2023.102421.
46. Hu, X.; Zhou, J.; Liu, G.; et al. Selection of microalgae for high CO_2 fixation efficiency and lipid accumulation from ten Chlorella strains using municipal wastewater. *J. Environ. Sci.* **2016**, *46*, 83–91. https://doi.org/10.1016/j.jes.2015.08.030.
47. Montoya-Vallejo, C.; Guzmán Duque, F.L.; Quintero Díaz, J.C. Biomass and lipid production by the native green microalgae *Chlorella sorokiniana* in response to nutrients, light intensity, and carbon dioxide: Experimental and modeling approach. *Front. Bioeng. Biotechnol.* **2023**, *11*, 1149762. https://doi.org/10.3389/fbioe.2023.1149762.
48. Gabrielyan, D.A.; Sinetova, M.A.; Savinykh, G.A.; et al. Productivity and Carbon Utilization of Three Green Microalgae Strains with High Biotechnological Potential Cultivated in Flat-Panel Photobioreactors. *Phycology* **2025**, *5*, 43. https://doi.org/10.3390/phycology5030043.
49. Naduthodi, M.I.S.; Claassens, N.J.; D'Adamo, S.; et al. Synthetic Biology Approaches to Enhance Microalgal Productivity. *Trends Biotechnol.* **2021**, *39*, 1019–1036. https://doi.org/10.1016/j.tibtech.2020.12.010.
50. Gupta, P.; Prabhakaran, R.; Soni, N.; et al. Utilization of Genetically Engineered RuBisCO Enzyme for Capturing Carbon Dioxide to Address Environmental Challenges-An Overview. *Water Air Soil Pollut.* **2026**, *237*, 58. https://doi.org/10.1007/s11270-025-08639-y.
51. Poudel, S.; Pike, D.H.; Raanan, H.; et al. Biophysical analysis of the structural evolution of substrate specificity in RuBisCO. *Proc. Natl. Acad. Sci. USA* **2020**, *117*, 30451–30457. https://doi.org/10.1073/pnas.2018939117.
52. Wei, L.; Wang, Q.; Xin, Y.; et al. Enhancing photosynthetic biomass productivity of industrial oleaginous microalgae by overexpression of RuBisCO activase. *Algal Res.* **2017**, *27*, 366–375. https://doi.org/10.1016/j.algal.2017.07.023.
53. Prywes, N.; Phillips, N.R.; Tuck, O.T.; et al. Rubisco Function, Evolution, and Engineering. *Annu. Rev. Biochem.* **2023**, *92*, 385–410. https://doi.org/10.1146/annurev-biochem-040320-101244.
54. Lin, M.T.; Hanson, M.R. Red algal Rubisco fails to accumulate in transplastomic tobacco expressing *Griffithsia monilis RbcL* and *RbcS* genes. *Plant Direct* **2018**, *2*, e00045. https://doi.org/10.1002/pld3.45.

55. Hennacy, J.H.; Jonikas, M.C. Prospects for Engineering Biophysical CO_2 Concentrating Mechanisms into Land Plants to Enhance Yields. *Annu. Rev. Plant Biol.* **2020**, *71*, 461–485. https://doi.org/10.1146/annurev-arplant-081519-040100.
56. Freeman Rosenzweig, E.S.; Xu, B.; Kuhn Cuellar, L.; et al. The Eukaryotic CO_2-Concentrating Organelle Is Liquid-like and Exhibits Dynamic Reorganization. *Cell* **2017**, *171*, 148–162.e19. https://doi.org/10.1016/j.cell.2017.08.008.
57. Mackinder, L.C.M.; Meyer, M.T.; Mettler-Altmann, T.; et al. A repeat protein links Rubisco to form the eukaryotic carbon-concentrating organelle. *Proc. Natl. Acad. Sci. USA* **2016**, *113*, 5958–5963. https://doi.org/10.1073/pnas.1522866113.
58. Kato, Y.; Oyama, T.; Inokuma, K.; et al. Enhancing carbohydrate repartitioning into lipid and carotenoid by disruption of microalgae starch debranching enzyme. *Commun. Biol.* **2021**, *4*, 450. https://doi.org/10.1038/s42003-021-01976-8.
59. Lee, H.J.; Choi, J.; Lee, S.M.; et al. Photosynthetic CO_2 Conversion to Fatty Acid Ethyl Esters (FAEEs) Using Engineered Cyanobacteria. *J. Agric. Food Chem.* **2017**, *65*, 1087–1092. https://doi.org/10.1021/acs.jafc.7b00002.
60. Huang, Z.; Zhang, J.; Pan, M.; et al. Valorisation of microalgae residues after lipid extraction: Pyrolysis characteristics for biofuel production. *Biochem. Eng. J.* **2022**, *179*, 108330. https://doi.org/10.1016/j.bej.2021.108330.
61. Liu, B.; Xing, Z.; Xue, Y.; et al. Effect of Pyrolysis Temperature on the Carbon Sequestration Capacity of Spent Mushroom Substrate Biochar in the Presence of Mineral Iron. *Molecules* **2024**, *29*, 5712. https://doi.org/10.3390/molecules29235712.
62. Bird, M.I.; Wurster, C.M.; de Paula Silva, P.H.; et al. Algal biochar—Production and properties. *Bioresour. Technol.* **2011**, *102*, 1886–1891. https://doi.org/10.1016/j.biortech.2010.07.106.
63. Leng, L.; Yang, L.; Chen, J.; et al. A review on pyrolysis of protein-rich biomass: Nitrogen transformation. *Bioresour. Technol.* **2020**, *315*, 123801. https://doi.org/10.1016/j.biortech.2020.123801.
64. Szyjka, S.J.; Mandal, S.; Schoepp, N.G.; et al. Evaluation of phenotype stability and ecological risk of a genetically engineered alga in open pond production. *Algal Res.* **2017**, *24*, 378–386. https://doi.org/10.1016/j.algal.2017.04.006.
65. Sebesta, J.; Xiong, W.; Guarnieri, M.T.; et al. Biocontainment of Genetically Engineered Algae. *Front. Plant Sci.* **2022**, *13*, 839446. https://doi.org/10.3389/fpls.2022.839446.
66. Béchet, Q.; Coulombier, N.; Vasseur, C.; et al. Full-scale validation of an algal productivity model including nitrogen limitation. *Algal Res.* **2018**, *31*, 377–386. https://doi.org/10.1016/j.algal.2018.02.010.
67. Qin, J.; Jia, M.; Sun, J. Examining the effects of elevated CO_2 on the growth kinetics of two microalgae, *Skeletonema dohrnii* (Bacillariophyceae) and *Heterosigma akashiwo* (Raphidophyceae). *Front. Mar. Sci.* **2024**, *11*, 1347029. https://doi.org/10.3389/fmars.2024.1347029.
68. Bošnjaković, M.; Sinaga, N. The Perspective of Large-Scale Production of Algae Biodiesel. *Appl. Sci.* **2020**, *10*, 8181. https://doi.org/10.3390/app10228181.
69. Novoveská, L.; Nielsen, S.L.; Eroldoğan, O.T.; et al. Overview and Challenges of Large-Scale Cultivation of Photosynthetic Microalgae and Cyanobacteria. *Mar. Drugs* **2023**, *21*, 445. https://doi.org/10.3390/md21080445.
70. Jerney, J.; Spilling, K. Large Scale Cultivation of Microalgae: Open and Closed Systems. In *Biofuels from Algae: Methods and Protocols*; Springer: New York, NY, USA, 2018; pp. 1–8. https://doi.org/10.1007/7651_2018_130.
71. Gurau, S.; Imran, M.; Ray, R.L. Algae: A cutting-edge solution for enhancing soil health and accelerating carbon sequestration—A review. *Environ. Technol. Innov.* **2025**, *37*, 103980. https://doi.org/10.1016/j.eti.2024.103980.
72. Banerjee, S.; Ramaswamy, S. Comparison of productivity and economic analysis of microalgae cultivation in open raceways and flat panel photobioreactor. *Bioresour. Technol. Rep.* **2019**, *8*, 100328. https://doi.org/10.1016/j.biteb.2019.100328.
73. McGowen, J.; Knoshaug, E.P.; Laurens, L.M.L.; et al. Outdoor annual algae productivity improvements at the pre-pilot scale through crop rotation and pond operational management strategies. *Algal Res.* **2023**, *70*, 102995. https://doi.org/10.1016/j.algal.2023.102995.
74. González-Camejo, J.; Viruela, A.; Ruano, M.V.; et al. Effect of light intensity, light duration and photoperiods in the performance of an outdoor photobioreactor for urban wastewater treatment. *Algal Res.* **2019**, *40*, 101511. https://doi.org/10.1016/j.algal.2019.101511.
75. Barahoei, M.; Hatamipour, M.S.; Afsharzadeh, S. CO_2 capturing by chlorella vulgaris in a bubble column photo-bioreactor; Effect of bubble size on CO_2 removal and growth rate. *J. CO_2 Util.* **2020**, *37*, 9–19. https://doi.org/10.1016/j.jcou.2019.11.023.
76. Penloglou, G.; Pavlou, A.; Kiparissides, C. Recent Advancements in Photo-Bioreactors for Microalgae Cultivation: A Brief Overview. *Processes* **2024**, *12*, 1104. https://doi.org/10.3390/pr12061104.
77. Dębowski, M.; Krzemieniewski, M.; Zieliński, M.; et al. Immobilized Microalgae-Based Photobioreactor for CO_2 Capture (IMC-CO2PBR): Efficiency Estimation, Technological Parameters, and Prototype Concept. *Atmosphere* **2021**, *12*, 1031. https://doi.org/10.3390/atmos12081031.
78. Clippinger, J.; Davis, R. *Techno-Economic Analysis for the Production of Algal Biomass via Closed Photobioreactors: Future Cost Potential Evaluated Across a Range of Cultivation System Designs*; Technical Report NREL/TP-5100-72716; National Renewable Energy Laboratory: Golden, CO, USA, 2019. Available online: https://docs.nlr.gov/docs/fy19osti/72716.pdf (accessed on 19 October 2026).

79. Skifa, I.; Chauchat, N.; Cocquet, P.H.; et al. Microalgae cultivation in raceway ponds: Advances, challenges, and hydrodynamic considerations. *EFB Bioecon. J.* **2025**, *5*, 100073. https://doi.org/10.1016/j.bioeco.2024.100073.
80. Quiroz, D.; Greene, J.M.; Limb, B.J.; et al. Global Life Cycle and Techno-Economic Assessment of Algal-Based Biofuels. *Environ. Sci. Technol.* **2023**, *57*, 11541–11551. https://doi.org/10.1021/acs.est.3c02892.
81. Loc, N.X.; Phuong, D.T.M. Optimizing biochar production: A review of recent progress in lignocellulosic biomass pyrolysis. *Front. Agric. Sci. Eng.* **2024**, *12*, 148–172. https://doi.org/10.15302/j-fase-2024597.
82. Chatterjee, R.; Sajjadi, B.; Chen, W.Y.; et al. Effect of Pyrolysis Temperature on PhysicoChemical Properties and Acoustic-Based Amination of Biochar for Efficient CO_2 Adsorption. *Front. Energy Res.* **2020**, *8*, 85. https://doi.org/10.3389/fenrg.2020.00085.
83. Bouaik, H.; Tabal, A.; Barakat, A.; et al. Optimal parameters and structural composition of bio-oil and biochar from intermediate pyrolysis of red algal biomass. *Comptes Rendus. Chim.* **2021**, *24*, 85–99. https://doi.org/10.5802/crchim.90.
84. Zou, G.; Luo, Y. Effects of temperature and heating rate on the characteristics of molded bio-char. *BioResources* **2020**, *15*, 2277–2292. https://doi.org/10.15376/biores.15.2.2277-2292.
85. Chen, J.; Wang, P.; Ding, L.; et al. The comparison study of multiple biochar stability assessment methods. *J. Anal. Appl. Pyrolysis* **2021**, *156*, 105070. https://doi.org/10.1016/j.jaap.2021.105070.
86. Wang, J.; Xiong, Z.; Kuzyakov, Y. Biochar stability in soil: Meta-analysis of decomposition and priming effects. *GCB Bioenergy* **2016**, *8*, 512–523. https://doi.org/10.1111/gcbb.12266.
87. Chiodo, V.; Zafarana, G.; Maisano, S.; et al. Pyrolysis of different biomass: Direct comparison among Posidonia Oceanica, Lacustrine Alga and White-Pine. *Fuel* **2016**, *164*, 220–227. https://doi.org/10.1016/j.fuel.2015.09.093.
88. An, X.; Zhu, Z.; Luo, X.; et al. Effects of Raw Materials and Pyrolysis Temperatures on Physicochemical Properties of Biochars Derived from Hemp Stalks. *Plants* **2025**, *14*, 2564. https://doi.org/10.3390/plants14162564.
89. Lou, B.; Parker, M.; Roumeli, E. Effects of pyrolysis temperature of macroalgal biomass on the structure and mechanical properties of produced biochar. *BioResources* **2025**, *20*, 4152–4173. https://doi.org/10.15376/biores.20.2.4152-4173.
90. Song, Y.; Tan, J.; Jin, M.; et al. Effect of pyrolysis temperature and heating rate on the physicochemical properties of alkali lignin-derived biochar: A comparative study of fast and slow pyrolysis. *J. Anal. Appl. Pyrolysis* **2025**, *191*, 107236. https://doi.org/10.1016/j.jaap.2025.107236.
91. Mondal, A.K.; Hinkley, C.; Krishnan, L.; et al. Macroalgae-based biochar: Preparation and characterization of physicochemical properties for potential applications. *RSC Sustain.* **2024**, *2*, 1828–1836. https://doi.org/10.1039/d4su00008k.
92. Michalak, I.; Baśladyńska, S.; Mokrzycki, J.; et al. Biochar from A Freshwater Macroalga as A Potential Biosorbent for Wastewater Treatment. *Water* **2019**, *11*, 1390. https://doi.org/10.3390/w11071390.
93. Ram, S.; Vasudev, V.; Ku, X. Characterization and kinetic analysis of lignocellulosic and algal biochar combustion. *Int. J. Fluid Eng.* **2024**, *1*, 024302. https://doi.org/10.1063/5.0194358.
94. Takal, D.; Howell, N.; Partheepan, J.; et al. Lab- and pilot-scale biochar production from cotton gin waste. *Clean. Eng. Technol.* **2026**, *32*, 101222. https://doi.org/10.1016/j.clet.2026.101222.
95. Ganesan, A.; Rezazgui, O.; Langlois, S.; et al. Pyrolytic conversion of construction, renovation, and demolition (CRD) wood wastes in Québec to biochar: Production, characterization, and identifying relevant stability indices for carbon sequestration. *Sci. Total Environ.* **2025**, *965*, 178650. https://doi.org/10.1016/j.scitotenv.2025.178650.
96. Chalivendra, S. Sustainable Biochar Production from Algal Biomass for Soil Enhancement. *Int. J. Sci. Res. Sci. Technol.* **2024**, *11*, 602–616. https://doi.org/10.32628/ijsrst241161113.
97. Niu, Q.; Du, X.; Xing, P.; et al. Two-step catalytic pyrolysis of microalgae combining in-bed CaO with vapour phase upgrading over Ni-ZSM-5/bentonite extrudates. *Green Chem.* **2025**, *27*, 13667–13681. https://doi.org/10.1039/d5gc03529e.
98. El Hosry, L.; Elias, V.; Chamoun, V.; et al. Maillard Reaction: Mechanism, Influencing Parameters, Advantages, Disadvantages, and Food Industrial Applications: A Review. *Foods* **2025**, *14*, 1881. https://doi.org/10.3390/foods14111881.
99. Durak, H.; Yarbay, R.Z.; Atilgan Türkmen, B. Hydrothermal Carbonization of Biomass for Hydrochar Production: Mechanisms, Process Parameters, and Sustainable Valorization. *Processes* **2026**, *14*, 339. https://doi.org/10.3390/pr14020339.
100. Libra, J.A.; Ro, K.S.; Kammann, C.; et al. Hydrothermal carbonization of biomass residuals: A comparative review of the chemistry, processes and applications of wet and dry pyrolysis. *Biofuels* **2011**, *2*, 71–106. https://doi.org/10.4155/bfs.10.81.
101. Heilmann, S.M.; Davis, H.T.; Jader, L.R.; et al. Hydrothermal carbonization of microalgae. *Biomass Bioenergy* **2010**, *34*, 875–882. https://doi.org/10.1016/j.biombioe.2010.01.032.
102. Verra. *Consultation: Major Revision to Biochar Methodology (VM0044)*; Verra: Washington, DC, USA, 2026. Available online: https://verra.org/consultation-major-revision-to-biochar-methodology-vm0044/ (accessed on *19 October* 2026).
103. Olalere, O.A.; Gan, C.Y.; Yap, P.G.; et al. Leveraging microalgae utilization for anthropogenic CO_2 emission abatement in Malaysian urban centre: Revisiting the sustainable development goal (SDG-13). *Phys. Chem. Earth Parts A/B/C* **2023**, *129*, 103356. https://doi.org/10.1016/j.pce.2022.103356.

104. Slegers, P.M.; Olivieri, G.; Breitmayer, E.; et al. Design of Value Chains for Microalgal Biorefinery at Industrial Scale: Process Integration and Techno-Economic Analysis. *Front. Bioeng. Biotechnol.* **2020**, *8*, 550758. https://doi.org/10.3389/fbioe.2020.550758.
105. Deepa, V.H.; Jayashree, D.R.; Pramod, P.D.; et al. Microalgae Based Air Purifiers: A Sustainable Approach to Urban Air Pollution Mitigation in Environmental Context. *Int. J. Environ. Sci.* **2025**, *11*, 7725–7729. https://doi.org/10.64252/6va6dy12.
106. Wiatrowski, M.; Klein, B.C.; Davis, R.W.; et al. Techno-economic assessment for the production of algal fuels and value-added products: Opportunities for high-protein microalgae conversion. *Biotechnol. Biofuels Bioprod.* **2022**, *15*, 8. https://doi.org/10.1186/s13068-021-02098-3.
107. Thakur, S.; Sahu, S.; Singh, G.; et al. Algal biomass biorefineries: Pretreatment strategies, conversion technologies, and multifunctional valorization for sustainable bioenergy and bioproducts. *Next Res.* **2026**, *9*, 101711. https://doi.org/10.1016/j.nexres.2026.101711.
108. Abdelfattah, A.; Ali, S.S.; Ramadan, H.; et al. Microalgae-based wastewater treatment: Mechanisms, challenges, recent advances, and future prospects. *Environ. Sci. Ecotechnol.* **2023**, *13*, 100205. https://doi.org/10.1016/j.ese.2022.100205.
109. Patel, A.K.; Tambat, V.S.; Chen, C.W.; et al. Recent advancements in astaxanthin production from microalgae: A review. *Bioresour. Technol.* **2022**, *364*, 128030. https://doi.org/10.1016/j.biortech.2022.128030.
110. Sung, Y.J.; Sim, S.J. Multifaceted strategies for economic production of microalgae *Haematococcus pluvialis*-derived astaxanthin via direct conversion of CO_2. *Bioresour. Technol.* **2022**, *344*, 126255. https://doi.org/10.1016/j.biortech.2021.126255.
111. Villaró, S.; Ciardi, M.; Morillas-España, A.; et al. Microalgae Derived Astaxanthin: Research and Consumer Trends and Industrial Use as Food. *Foods* **2021**, *10*, 2303. https://doi.org/10.3390/foods10102303.
112. De Silva, A.G.S.D.; Hashim, Z.K.; Solomon, W.; et al. Unveiling the Role of Edaphic Microalgae in Soil Carbon Sequestration: Potential for Agricultural Inoculants in Climate Change Mitigation. *Agriculture* **2024**, *14*, 2065. https://doi.org/10.3390/agriculture14112065.
113. Maghsoodi, M.R.; Hosseini, H.; Ghodszad, L.; et al. Nutrient recovery in pyrolysis biochars for agricultural applications–Biochemical compositions and nutrient extraction techniques. *Resour. Conserv. Recycl.* **2025**, *218*, 108237. https://doi.org/10.1016/j.resconrec.2025.108237.
114. Truong, H.B.; Dang, V.D.; Khedulkar, A.P.; et al. Biochar for pollution mitigation and renewable energy applications toward sustainability development. *RSC Adv.* **2026**, *16*, 5834–5851. https://doi.org/10.1039/d5ra09050d.
115. Zhang, Y.; He, M.; Wang, L.; et al. Biochar as construction materials for achieving carbon neutrality. *Biochar* **2022**, *4*, 59. https://doi.org/10.1007/s42773-022-00182-x.
116. IPCC. Appendix 4: Method for estimating the change in mineral soil organic carbon stocks from biochar amendments. In *2019 Refinement to the 2006 IPCC Guidelines for National Greenhouse Gas Inventories*; IPCC: Geneva, Switzerland, 2019; Volume 4, Chapter 2. Available online: https://www.ipcc-nggip.iges.or.jp/public/2019rf/pdf/4_Volume4/19R_V4_Ch02_Ap4_Biochar.pdf (accessed on 19 October 2026).
117. Cayuela, M.L.; van Zwieten, L.; Singh, B.P.; et al. Biochar's role in mitigating soil nitrous oxide emissions: A review and meta-analysis. *Agric. Ecosyst. Environ.* **2014**, *191*, 5–16. https://doi.org/10.1016/j.agee.2013.10.009.
118. Borchard, N.; Schirrmann, M.; Cayuela, M.L.; et al. Biochar, soil and land-use interactions that reduce nitrate leaching and N2O emissions: A meta-analysis. *Sci. Total Environ.* **2019**, *651*, 2354–2364. https://doi.org/10.1016/j.scitotenv.2018.10.060.
119. European Commission. Commission Delegated Regulation (EU) 2021/2088 of 7 July 2021 Amending Annexes II, III and IV to Regulation (EU) 2019/1009 of the European Parliament and of the Council for the Purpose of Adding Pyrolysis and Gasification Materials as a Component Material Category in EU Fertilising Products (Text with EEA relevance). 2021. Available online: https://eur-lex.europa.eu/eli/reg_del/2021/2088/oj/eng (accessed on 19 October 2026).
120. European Parliament and Council of the European Union. Regulation (EU) 2019/1009 of the European Parliament and of the Council of 5 June 2019 Laying Down Rules on the Making Available on the Market of EU Fertilising Products and Amending Regulations (EC) No 1069/2009 and (EC) No 1107/2009 and Repealing Regulation (EC) No 2003/2003 (Text with EEA relevance). 2019. Available online: https://eur-lex.europa.eu/eli/reg/2019/1009/oj/eng (accessed on 19 October 2026).
121. Isometric. *Biochar Production and Storage (v1.3)*; Isometric: London, UK, 2025. Available online: https://registry.isometric.com/protocol/biochar/1.3 (accessed on *19 October* 2026).
122. Moni, S.M.; Mahmud, R.; High, K.; et al. Life cycle assessment of emerging technologies: A review. *J. Ind. Ecol.* **2020**, *24*, 52–63. https://doi.org/10.1111/jiec.12965.
123. Peña, C.; Civit, B.; Gallego-Schmid, A.; et al. Using life cycle assessment to achieve a circular economy. *Int. J. Life Cycle Assess.* **2021**, *26*, 215–220. https://doi.org/10.1007/s11367-020-01856-z.
124. Liu, Y.; Xia, R.; Wang, Y.; et al. Life cycle environmental impact assessment of microalgae biodiesel derived from different processes. *Ind. Crops Prod.* **2025**, *233*, 121381. https://doi.org/10.1016/j.indcrop.2025.121381.

125. Moradiya, K.K.; Marathe, K.V. Life cycle assessment (LCA) of marine microalgae cultivation and harvesting process for the Indian context. *Sustain. Energy Technol. Assess.* **2023**, *56*, 103063. https://doi.org/10.1016/j.seta.2023.103063.
126. Madhu, K.; Pauliuk, S.; Dhathri, S.; et al. Understanding environmental trade-offs and resource demand of direct air capture technologies through comparative life-cycle assessment. *Nat. Energy* **2021**, *6*, 1035–1044. https://doi.org/10.1038/s41560-021-00922-6.
127. Mu, D.; Min, M.; Krohn, B.; et al. Life Cycle Environmental Impacts of Wastewater-Based Algal Biofuels. *Environ. Sci. Technol.* **2014**, *48*, 11696–11704. https://doi.org/10.1021/es5027689.
128. Kumar, P.K.; Vijaya Krishna, S.; Verma, K.; et al. Phycoremediation of sewage wastewater and industrial flue gases for biomass generation from microalgae. *S. Afr. J. Chem. Eng.* **2018**, *25*, 133–146. https://doi.org/10.1016/j.sajce.2018.04.006.
129. Singh, U.; Naaz, F.; Hawkins, T.R.; et al. Direct flue gas capture for algae cultivation and subsequent valorization: Evaluating life cycle emissions and costs. *Sustain. Energy Fuels* **2025**, *9*, 4392–4403. https://doi.org/10.1039/d5se00329f.
130. Amponsah, L.; Jones, E.; Davies, J.; et al. Life cycle assessment and techno-economic analysis of a macroalgal-derived biopolymer film coupled with carbon capture. *Sustain. Sci. Technol.* **2026**, *3*, 024001. https://doi.org/10.1088/2977-3504/ae4806.
131. Kane, S.; Bin Thaneya, A.; Gursel, A.P.; et al. Uncertainty in determining carbon dioxide removal potential of biochar. *Environ. Res. Lett.* **2025**, *20*, 014062. https://doi.org/10.1088/1748-9326/ad99e9.
132. Chisti, Y. Biodiesel from microalgae. *Biotechnol. Adv.* **2007**, *25*, 294–306. https://doi.org/10.1016/j.biotechadv.2007.02.001.
133. IPCC. Chapter 2: Stationary combustion. In *2006 IPCC Guidelines for National Greenhouse Gas Inventories*; IGES: Hayama, Japan, 2006; Volume 2.
134. Greene, J.M.; Quiroz, D.; Limb, B.J.; et al. Geographically-Resolved Techno-Economic and Life Cycle Assessment Comparing Microalgae-Based Renewable Diesel and Sustainable Aviation Fuel in the United States. *Environ. Sci. Technol.* **2025**, *59*, 3472–3483. https://doi.org/10.1021/acs.est.4c06742.
135. Nagireddi, S.; Agarwal, J.R.; Vedapuri, D. Carbon Dioxide Capture, Utilization, and Sequestration: Current Status, Challenges, and Future Prospects for Global Decarbonization. *ACS Eng. Au* **2024**, *4*, 22–48. https://doi.org/10.1021/acsengineeringau.3c00049.
136. Schenuit, F.; Gidden, M.J.; Boettcher, M.; et al. Secure robust carbon dioxide removal policy through credible certification. *Commun. Earth Environ.* **2023**, *4*, 349. https://doi.org/10.1038/s43247-023-01014-x.
137. Puro.earth. *Puro Biochar Methodology*, Edition 2025 (Version 2); Puro.earth: Helsinki, Finland, 2025. Available online: https://puro.earth/document-library (accessed on *19 October* 2026).
138. Puro.earth. *Preliminary Assessment Public Summary (Applicable Methodology: Biochar, Edition 2022, V3)*; Puro.earth: Helsinki, Finland, 2025. Available online: https://my.puro.earth/supplier-listing/preliminary-assessment/31bb116b-1df7-41f3-8c52-f5c4d3ea4f2d (accessed on *19 October* 2026).
139. Zhou, X.; Li, J.; Zhao, X.; et al. Resource recovery in life cycle assessment of sludge treatment: Contribution, sensitivity, and uncertainty. *Sci. Total Environ.* **2022**, *806*, 150409. https://doi.org/10.1016/j.scitotenv.2021.150409.
140. Mohamed, B.A.; Ruan, R.; Bilal, M.; et al. Co-pyrolysis of sewage sludge and biomass for stabilizing heavy metals and reducing biochar toxicity: A review. *Environ. Chem. Lett.* **2023**, *21*, 1231–1250. https://doi.org/10.1007/s10311-022-01542-6.
141. DeAngelo, J.; Saenz, B.T.; Arzeno-Soltero, I.B.; et al. Economic and biophysical limits to seaweed farming for climate change mitigation. *Nat. Plants* **2023**, *9*, 45–57. https://doi.org/10.1038/s41477-022-01305-9. Correction in *Nat. Plants* **2023**, *9*, 674.
142. FAO. *Inorganic Fertilizers 1990–2021*; FAOSTAT Analytical Brief Series No. 68; Food and Agriculture Organization of the United Nations: Rome, Italy, 2023. Available online: https://www.fao.org/3/cc6823en/cc6823en.pdf (accessed on 19 October 2026).