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The True Footprint of Clean Energy: A Life-Cycle Comparison of Solar, Wind, and Microalgae Biofuel

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ABSTRACT: As the global climate crisis escalates, a transition to sustainable energy is vital for curbing greenhouse gas emissions. This paper provides a comparative Life-Cycle Assessment (LCA) to evaluate the environmental impacts of three renewable technologies: solar photovoltaic (PV) systems, wind energy, and microalgae biofuel. Synthesizing data from 25 selected peer-reviewed studies and organizational reports, this analysis quantifies key environmental metrics, including Global Warming Potential (GWP), resource demands, and Energy Payback Time (EPBT). The findings reveal significant variability. Modern onshore wind turbines exhibit the lowest consistent emissions (as low as 5.6 g CO2-eq/kWh), followed by solar PV systems (19–50 g CO2-eq/kWh), while microalgae biofuel displays the widest range, from being a net carbon sink in optimized systems (-48 to -623 g CO2-eq/kWh) to a major emitter in others. These results show that although onshore wind has the lowest mean life-cycle emissions, the optimal choice is highly context-dependent, determined by regional climate, technological maturity, and logistical factors. This underscores the need for sophisticated, context-specific energy policies to achieve a sustainable future.

KEYWORDS: Energy: Physical, Sustainable Design, Life-Cycle Assessment, Solar Photovoltaics, Microalgae Biofuel.

Introduction

Mitigating climate change requires a global transition from fossil fuels, which are responsible for approximately 80% of energy-related CO_2 emissions. Solar photovoltaic (PV) systems, wind energy, and microalgae biofuel represent diverse pathways to decarbonize energy production, each with distinct mechanisms and environmental implications. Solar PV systems convert photons from sunlight into electricity via semiconductor materials, a process known as the photovoltaic effect. Wind energy captures kinetic energy from air movement using turbines, which drive generators to produce electricity. Microalgae biofuel utilizes photosynthetic microorganisms to produce biomass, subsequently processed into liquid fuels such as biodiesel or biobutanol. Although these technologies have low operational emissions, their true sustainability can only be understood by examining their entire life cycle, from raw material extraction to end-of-life management.

Evaluating the true sustainability of these technologies requires a method known as Life-Cycle Assessment (LCA), governed by the ISO 14040 series of standards.[1] LCA allows for a "cradle-to-grave" approach. This method assesses environmental impacts from raw material extraction to the End of Life (EOL) stage, where products are recycled or disposed of, influencing waste management and resource recovery.[2] This assessment quantifies various impacts, including the emission of Greenhouse Gases (GHG). These gases, such as carbon dioxide (CO_2) and methane, contribute to global warming.[3] To standardize these impacts for consistent comparison, emissions are often converted to a CO_2 Equivalence (CO_2-eq), a metric based on each gas's Global Warming Potential (GWP) as defined by the Intergovernmental Panel on Climate Change (IPCC).[4] This metric highlights a significant contrast to traditional fossil fuels. For instance, the GWP of electricity from hard coal is over 99% higher than that of clean sources like hydropower.[3] Furthermore, LCA evaluates energy efficiency through metrics like the Energy Payback Time (EPBT), which measures the time a system needs to generate the energy consumed during its own production.[5] However, it is important to note the methodological challenges inherent in LCA studies; variations in system boundaries and impact assessment methods, which categorize environmental effects differently (e.g., CML 2001, ReCiPe), can complicate direct comparisons across studies.[6]

While solar PV and wind energy are established renewable technologies, microalgae biofuel remains a less commercially developed but promising alternative with the unique potential for direct CO_2 sequestration. However, its viability is often limited by energy-intensive cultivation and processing steps. To properly contextualize its potential, a direct comparison with mature renewable technologies is needed. Therefore, this review synthesizes current LCA research to compare the carbon footprints, measured in grams of CO_2 equivalent per kilowatt-hour of energy generated (g CO_2-eq/kWh), alongside resource demands, and energy efficiency of microalgae biofuel, alongside solar PV and wind energy. Accordingly, this paper evaluates the distinct environmental trade-offs of each technology to demonstrate that their sustainability is highly context-dependent, thereby providing a foundation for more effective, region-specific energy policies.

This review first evaluates solar PV systems, followed by wind turbines, and then microalgae biofuel, analyzing each through the four key LCA phases: material cultivation and processing; production and manufacturing; operation; and end-of-life management. This final phase evaluates the impacts of disposal and recycling, where resource recovery can yield environmental credits that offset upstream emissions. Following

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these individual assessments, a comparative analysis evaluates the distinct environmental trade-offs of each technology, and the review concludes by synthesizing these findings to provide a foundation for more effective, region-specific energy policies.

■ Methods

To collect sources, a search was conducted primarily using Google Scholar for peer-reviewed studies. Moreover, this was supplemented by a review of "grey literature," including technical reports from industry and governmental organizations (e.g., U.S. Environmental Protection Agency, Vestas). The search used keyword combinations such as "Life Cycle Assessment," "LCA," "solar PV," "wind power," "microalgae biofuel," "GWP," "GHG," and "Energy Payback Time (EPBT)."

To keep the data relevant to current technologies, the review focused on studies published within the last 15 years (2010-2025). Priority was given to papers that provided detailed, quantitative, life-cycle stage data and included comprehensive figures or tables. Additionally, both primary LCA studies and existing meta-analyses were included. Studies were excluded if they were not published in English or were not directly relevant to the metrics for core comparison.

Key data points for Global Warming Potential (GWP), Greenhouse Gas (GHG) Emissions, and Energy Payback Time (EPBT) were manually extracted from the selected literature. This quantitative data was then compiled to create the comparative analysis presented in the paper's main table and Figures 1 to 3.

■ Results of Literature Search

The search and screening process described in the Methods section yielded 25 LCA studies that were selected for the final quantitative and qualitative synthesis.

The final papers used for analysis, comparison, and data extraction consisted of 6 studies for solar PV, 9 studies for wind energy, and 10 studies for microalgae biofuel. The paper's bibliography is larger than this total, as it also includes foundational texts, methodological standards (such as ISO 14040), and reports used for contextual data.

■ Discussion

Life-Cycle Analysis of Photovoltaic (PV) Systems: Overview and Scope:

A solar photovoltaic (PV) system functions by directly converting energy from sunlight into electricity. This process, known as the photovoltaic effect, occurs within solar panels, which are the visible component of a system. These panels are typically made from many individual solar cells, made from semiconductor materials like silicon, which absorb photons and release electrons to create direct current (DC) electricity.[6]

However, the life cycle of a complete system includes several other vital components grouped under the Balance of System (BoS).[6] For example, inverters are required to convert the direct current (DC) electricity produced by the panels into alternating current (AC) usable by the grid or in a home.[6] Other components included in the BoS are the mounting structures, which provide physical support, as well as necessary wiring, connectors, and safety equipment.

This section evaluates the GWP of solar PV systems, and the analysis covers three generations of PV technology: first-generation crystalline silicon (mono-Si and multi-Si); second-generation thin-films such as amorphous silicon (a-Si), cadmium telluride (CdTe), and copper indium gallium selenide (CIGS); and third-generation emerging technologies, including organic photovoltaics (OPV), perovskite solar cells (PSC), and quantum dot (QD) PV. However, it is important to note that these emerging technologies have not yet achieved large-scale commercial deployment. While distinct, for this review, mono-Si and multi-Si will often be discussed collectively as 'crystalline silicon' systems to facilitate a clearer comparison across generations. Lifecycle assessments of PV systems report a wide range of environmental impacts, with GWP values ranging from 2.89 to 150 g CO_2-eq/kWh and an EPBT of 0.24 to 6.0 years, largely dependent on technology type, system design, and solar insolation.[6]

Material Cultivation and Processing:

The life cycle of a PV system begins with the extraction and processing of raw materials, a phase that often covers a significant portion of its total environmental footprint, with impacts varying distinctly across technology generations. For first-generation systems, key materials include silicon, aluminum, and steel. The purification of silicon is a highly energy-intensive process. It transforms metallurgical-grade silicon into solar-grade silicon and requires extremely high temperatures to achieve the necessary purity. This single step accounts for 52.4% of the environmental impacts for multi-crystalline silicon systems.[6] Aluminum and steel, used for frames and supports, increase ozone layer depletion potential, which refers to the degradation of stratospheric ozone.[6]

Second-generation systems utilize materials such as cadmium telluride, copper indium gallium selenide, amorphous silicon, glass, and aluminum. Processing these materials is less energy-intensive than first-generation technologies, but the use of fluorinated gases, such as sulfur hexafluoride, in manufacturing increases greenhouse gas emissions.[5] However, recovery of glass and aluminum during processing helps mitigate environmental impacts.[6] One study found that the GWP for cadmium telluride systems ranges from 19 to 30 g CO_2-eq/kWh.[5]

Third-generation systems incorporate materials like fullerenes, which have a high embodied energy of 35 to 123 GJ/kg.[6] This value is dramatically higher than the 1.96 to 5.85 GJ/kg required for common polymers. To put that into perspective, the energy needed to create just one kilogram of fullerenes could power an average U.S. home for nearly a full year at the low end (35 GJ) and over three years at the high end (123 GJ). These findings emphasize the significant energy investment needed for these advanced materials.[6] The GWP for QD-PV can be as low as 2.89 g CO_2-eq/kWh, demonstrating its potential for very low lifecycle emissions; however, this environmental advantage is offset by significant concerns regarding heavy metal emissions from its components.[6]

Production and Manufacturing:

Similar to material cultivation, the production and manufacturing phase is a major contributor to the environmental impacts of PV systems due to energy-intensive processes.[6] For first-generation systems, manufacturing involves two primary methods: the Czochralski process, which grows single-crystal silicon ingots by pulling a seed crystal from molten silicon, and casting, where molten silicon is cooled in a mold to form multi-crystalline silicon.[6] Both of these methods consume significant energy, leading to increased carbon dioxide emissions.[6] Multi-crystalline silicon systems have a cumulative energy demand of 1619 MJ/m^2.[6]

Second-generation systems employ thin-film deposition techniques, such as sputtering, which deposits material via plasma, or chemical vapor deposition, which deposits material from a gas phase.[5] These processes use less energy than first-generation methods, resulting in lower emissions overall; yet, they involve toxic materials like cadmium.[5]

Third-generation systems utilize roll-to-roll manufacturing, a continuous printing process for flexible substrates like organic PV, which reduces energy consumption.[6] Emissions vary due to diverse material choices and system scenarios. However, for a representative rooftop solar installation, OPV systems indicate a cumulative energy demand of 122 to 125 MJ/m^2.[6]

Operational Phase:

The operational phase of PV systems generates electricity with no direct emissions, making it zero direct emissions during operation.[6] PV systems produce negligible emissions during operation, effectively replacing fossil fuels, with impacts dependent on system efficiency, lifetime, and the amount of solar radiation received (insolation).[6] For first-generation systems, the EPBT ranges from 2.2 to 6.0 years, with multi-crystalline silicon systems specifically reported between 2.4 and 3.1 years, influenced by annual insolation levels between 1166.[7] and 1855.[6] peak sunshine hours.[6] Second-generation systems, such as cadmium telluride, have a shorter EPBT of 0.8 to 3.6 years.[5] Third-generation systems, like organic PV, achieve an EPBT of 0.24 to 1.2 years.[6] To illustrate the impact of performance improvements, one study found that PV systems generally increased in efficiency from 6% to 13%, and extending the lifetime from 15 to 30 years reduces the GWP from 38.[7] to 9.6 g CO_2-eq/kWh.[6] Geographical location impacts energy output, with regions like Northeast Nigeria producing 31% more energy compared to an average of 1700 kWh/m^2/year.[6]

End of Life Management:

End-of-life management for PV systems is critical, though current practices vary. Globally, landfilling remains the most common disposal method for both first- and second-generation panels, primarily due to economic factors and a lack of widespread recycling infrastructure.[7] However, environmentally superior recycling processes exist. For first-generation systems, recycling involves dismantling modules to recover glass (70% of mass), aluminum (10–18%), and ethylene-vinyl acetate (2–5%), with silicon crushed for raw material reuse rather than refurbishing entire panels.[6] Recycling 1000 kg of mono-crystalline silicon waste reduces the cumulative energy demand from 3150 MJ to 2780 MJ and decreases GWP by approximately 30% compared to landfilling the panels.[6]

For second-generation systems, such as CdTe, established recycling programs achieve high glass recovery (95% of mass) and recover toxic materials like cadmium, reducing GWP and non-renewable energy use.[5] Recycling these systems outperforms incineration in most environmental impact categories.[5] Third-generation systems face challenges due to limited recycling data, with studies often assuming landfilling.[6] However, glass and metal recovery reduces resource consumption, and polymer incineration offers energy recovery for these emerging technologies.[6]

Synthesis and Implications:

LCA studies underscore the sustainability of PV systems, with distinct characteristics across technology generations. As illustrated in **Figure 1**, the lifecycle GWP is dominated by the upstream phases, with material cultivation and fabrication accounting for the largest share of emissions. First-generation systems, such as crystalline silicon, exhibit a high EPBT of 2.2 to 6.0 years and a GWP of 49.9 g CO_2-eq/kWh, driven by energy-intensive production processes.[8] Second-generation systems, like cadmium telluride, achieve a moderate EPBT of 0.8 to 3.6 years and a GWP of 19 to 30 g CO_2-eq/kWh.[5] Third-generation systems, such as organic PV and QD-PV, offer a lower EPBT of 0.24 to 1.2 years but have a variable GWP of 2.89 to 150 g CO2-eq/kWh, as these technologies are still largely in the research and development (R&D) phase with significant concerns about material toxicity.[6]

Beyond the inherent properties of each generation, the life-cycle performance of any PV system is significantly modulated by several external factors. Improvements in system efficiency offer another path to lower impacts, with studies indicating GWP can be reduced by up to 30% through such advancements.[6] Geographical location affects EPBT, with insolation levels between 772.2 and 2100 peak sunshine hours leading to EPBT of 2.2 to 6.0 years.[6] The balance of system (BoS) components, such as inverters and mounting structures, can contribute up to 45% of total environmental impacts, stemming from the manufacturing of aluminum or steel mounts and the energy losses from inverters. Notably, these impacts can be mitigated by Building-integrated PV (BIPV) systems. In BIPV systems, panels also serve as structural elements like roof tiles or facades. This reduces the need for separate mounting hardware and displacing conventional building materials.[6] Longer lifetimes, typically 25 to 30 years for first-generation systems, enhance overall sustainability.[6] Effective EoL management through recycling provides a crucial opportunity to mitigate a portion of the initial environmental burden, with studies suggesting GWP reductions of up to 30% by avoiding the need for virgin raw material extraction.[7]

Looking ahead, it is important to acknowledge that limited data for third-generation technologies further hinders comprehensive assessments.[6] Nevertheless, PV systems remain promising, and improvements in efficiency, production processes, and recycling infrastructure are required to enhance

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their sustainability.[8] Future research should focus on mitigating risks from toxic material releases and developing dynamic LCA models to account for evolving emission factors.[6]

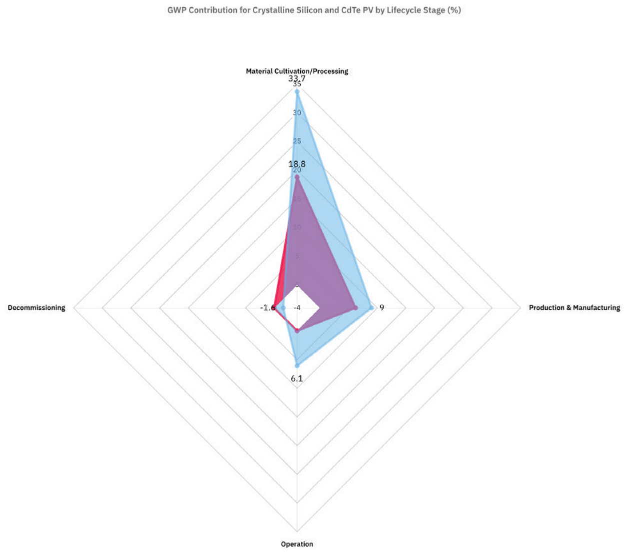

Figure 1: GWP contribution for crystalline silicon and CdTe PV by life-cycle stage (%). This radar chart compares the GWP breakdown for first-generation crystalline silicon (blue) and second-generation CdTe thin-film (red) solar PV. The distance from the center along each axis represents the percentage contribution of that stage to the total GWP. For example, Material Cultivation/Processing accounts for 33.7% of the GWP for crystalline silicon. Negative values, such as those for decommissioning, represent environmental credits from recycling that offset a portion of the emissions. The data shows that material cultivation and processing represent the largest GWP contribution for both technologies, especially for crystalline silicon. This highlights how the energy intensity of early stages, rather than manufacturing or operation, is the largest contributor to solar PV's carbon footprint. Data synthesized from Nugent & Sovacool (2014).[8]

Life-Cycle Analysis of Wind Energy: Overview and Scope:

Wind energy systems capture the kinetic energy of moving air and convert it into electricity. The main component is the turbine, which consists of several key parts. The rotor, which includes the large blades (often made of fiberglass composites), catches the wind and begins to turn. This rotation runs a generator, which is set inside the nacelle, the casing at the top of the turbine. Consequently, the generator converts the mechanical energy into electrical energy.[9]

Furthermore, this entire assembly is mounted on a tall tower, typically made of steel, to access stronger and more consistent winds. Finally, the large turbine is secured by a massive foundation, usually made of steel and concrete, which is a major contributor to the system's "cradle-to-grave" environmental footprint.[9]

This section evaluates the GWP of wind energy, which converts atmospheric kinetic energy into electricity via turbine-generator systems. The analysis covers both onshore turbines, which are the most common type, and offshore turbines, which are situated in bodies of water to access stronger, more consistent winds. Wind energy is a low-carbon technology, with an average lifecycle GWP of 34.1 g CO_2-eq/kWh, though this can range from 0.4 to 364.8 g CO_2-eq/kWh due to variations in turbine design, location, and methodological assumptions.[8] The EPBT for wind technologies is typically less than one year, highlighting their efficiency in offsetting embodied energy.[9] However, offshore wind generally exhibits higher GWP than onshore due to increased material and operational demands.

Material Cultivation and Processing:

This stage involves extracting and producing raw materials, which significantly contributes to GWP. The primary GWP contributors are steel and concrete due to the large quantities required; steel and iron together comprise 79.4% of offshore and 22.3% of onshore infrastructure.[9] Other materials with significant impacts include copper, aluminum, epoxy, and glass fiber. Moreover, steel production emissions vary by region (e.g., 1.97 t CO_2-eq/t in China versus 0.96 t CO_2-eq/t in the US), while copper in cables and epoxy in blades are also impactful, with epoxy being 48 times more GWP-intensive per kg than concrete.[9,10] Additionally, the electricity mix used in processing affects GWP, as low-carbon mixes reduce emissions compared to coal-heavy mixes. Therefore, for both onshore and offshore wind, this stage plays a dominant role in climate change impacts, accounting for over 79% onshore and 70% offshore.[9] Furthermore, offshore wind's higher GWP is exacerbated by larger high-impact material requirements for capital infrastructure, such as foundations and floaters.[9] The GWP for the material cultivation and processing stage of wind energy systems is reported to range from 2.3 to 26.6 g CO_2-eq/kWh.[11,12] It is worth noting that this range is narrower and generally lower than the one found in older, broader meta-analyses, which likely reflects ongoing technological improvements in turbine manufacturing and efficiency. This variability is attributed to factors such as turbine size and foundation type. The literature consistently identifies two primary drivers of this impact: material selection, which includes the large quantities of steel and concrete required, and the carbon intensity of the energy supply for manufacturing.

Production and Manufacturing (Construction):

While the precise contribution of each life-cycle phase varies between studies, one meta-analysis determined that the upstream phases are the most significant. The 'cultivation and fabrication' stage contributed approximately 71% of total emissions, with a mean GWP of 42.98 g CO_2-eq/kWh (range: 0.15–286.02), while the 'construction' stage contributed another 24%, with a mean GWP of 14.43 g CO_2-eq/kWh (range: 0.15–78.85).[8]

For example, specific case studies of low-emission turbines highlight the dominance of the manufacturing phase. The Vestas onshore V136-4.2 MW turbine has a total lifecycle GWP of 5.6 g CO_2-eq/kWh; its manufacturing phase contributes 8.8 g CO_2-eq/kWh, an impact that is significantly offset by end-of-life recycling credits.[13] A similar pattern is seen in the offshore V236-15 MW turbine, which has a total GWP of 7.0 g CO_2-eq/kWh, with manufacturing contributing 7.1 g CO_2-eq/kWh.[14] Consequently, offshore wind manufacturing has a higher GWP due to increased material inputs, particularly for foundations (16% of Acidification Potential, correlated with

GWP) and site cables (11%).[14] As another specific example case, in floating offshore wind farms like Hywind Tampen, manufacturing contributes 26.79% to a total GWP of 36.78 g CO_2-eq/kWh, whereas in fixed offshore farms like Dogger Bank, it contributes 43.84% to a total GWP of 26.15 g CO_2-eq/kWh.[15] Furthermore, component-specific GWP for offshore turbines includes blades (21%), gear and mainshaft (18%), and nacelle (10%).[14] In contrast, onshore wind manufacturing, dominated by towers and cables, has lower GWP due to simpler foundations and less cabling.[13] Notably, larger turbines with advanced direct drive generator technology, such as 3.2 MW onshore and 6.0 MW offshore models, perform better than smaller geared ones in terms of GWP.[9]

This highlights a key distinction: the direct operations at the manufacturing company itself (e.g., factory energy use) contribute less than 1% to total life-cycle impacts. The vast majority of the manufacturing footprint is attributed to the embodied energy in the raw materials, or 'upstream material production'.[9] Transport contributes minimally (0.05–2.52%), indicating that material production primarily drives GWP differences.[10]

Operational Phase:

According to the meta-analysis by Nugent *et al*. discussed in Section 3.3, the operational phase (covering operation, maintenance, and servicing) contributes approximately 24% of the total GWP.[8] The mean GWP for this phase was reported as 14.36 g CO_2-eq/kWh, with a range of 0.02 to 83.6 g CO_2-eq/kWh. However, this value is derived from a smaller subset of studies than the overall lifecycle average. This research limitation explains why the percentage is not a direct calculation of the two means.[8] For the Vestas onshore turbine, this phase is minor compared to manufacturing, thus contributing marginally to the total GWP of 5.6 g CO_2-eq/kWh, primarily through servicing and replacement parts.[13] However, offshore wind operations are more GWP-intensive, contributing 1.19 g CO_2-eq/kWh to the Vestas offshore turbine's GWP. This is driven by the significant fuel consumption of support vessels, such as Service Operation Vessels (SOVs) using 170 liters per hour (l/h) and tugboats using 596 l/h.[10,14] To put that into context, a single tugboat at full operation consumes the equivalent of about 12 full car fuel tanks (50L) every hour. Additionally, this phase's contribution to environmental impacts for offshore turbines varies widely, ranging from less than 1% to 41%, depending on the impact category.[14] Moreover, for floating offshore wind (Hywind Tampen), fuel for Operations and Maintenance (O&M) vessels (specialized ships that service and repair offshore wind farms) contributes 5.98 g CO_2-eq/kWh and spare parts 5.22 g CO_2-eq/kWh, while for fixed offshore wind (Dogger Bank), these values are 3.82 g CO_2-eq/kWh and 5.06 g CO_2-eq/kWh, respectively.[15] Consequently, the enhanced offshore GWP results from extended vessel travel distances and failure rates (e.g., mooring chains at 0.148 failures/year).[15] Additionally, turbine lifespan significantly affects GWP, with longer lifespans (30 years offshore vs. 20 onshore) reducing per-kWh GWP.[13,14] Furthermore, this phase highlights the shared responsibility across multiple stakeholders in the value chain, emphasizing the need for coordinated environmental management across stakeholders.[9]

End-of-Life Management:

The average lifespan of a wind turbine is typically between 20 and 30 years, after which the end-of-life management stage, involving decommissioning, dismantling, recycling, and disposal, begins. This stage often contributes to GWP credits, offsetting 19.4% of lifecycle emissions, with a relative mean credit of -11.64 g CO_2-eq/kWh (range: -59.4 to 0.5 g CO_2-eq/kWh).[8] It is important to clarify that these GWP credits do not represent direct carbon sequestration from the atmosphere. Rather, they are an accounting mechanism in LCA that reflects the avoided emissions that would have resulted from producing virgin raw materials. Recycling components like steel averts the energy and GWP associated with mining iron ore and manufacturing new steel. Consequently, this environmental benefit is then credited back to the turbine's life cycle. Moreover, effective end-of-life treatment can lead to significant savings, reducing climate change impacts by approximately 20–30% through recycling.[9] For instance, for the Vestas onshore turbine, effective recycling is a key factor in its low environmental impact; the total lifecycle GWP for this turbine is 5.6 g CO_2-eq/kWh, a figure achieved in part through high metal recycling rates (e.g., 98% for foundations, 95% for major components).[13] Similarly, the offshore Vestas turbine benefits, with recycling credits lowering the total GWP of 7.0 g CO_2-eq/kWh, assuming 90% recovery for metals like steel and copper, though only 50% for foundations and cables.[14] In floating offshore wind farms, the direct emissions from the decommissioning process itself are minimal (up to 5% of impacts), with GWP impacts of 2.17 g CO_2-eq/kWh, compared to 1.16 g CO_2-eq/kWh for fixed offshore farms.[15] However, a significant challenge in this phase is the disposal of turbine blades. The composite materials from which they are made, typically fiberglass and epoxy resins, are inherently difficult to separate, which makes them economically and technically challenging to recycle with current technologies. As a result, most blades are sent to landfills, representing a growing waste management concern.[16]

Synthesis and Implications:

Wind energy exhibits a low lifecycle GWP, with a performance range for modern turbines between 5.6 and 36.78 g CO_2-eq/kWh, depending on the specific technology and location.[13,14] The distribution of these impacts across the lifecycle, as shown in **Figure 2**, highlights that onshore wind generally outperforms offshore wind due to lower material and operational demands. Specifically, the production and manufacturing stage dominates GWP for both, influenced greatly by steel and concrete, while offshore wind experiences higher impacts from complex foundations and cables.[15] In contrast, operational GWP is minimal onshore but more significant offshore, contributing 1.19 g CO_2-eq/kWh for the offshore V236-15 MW turbine due to SOV fuel consumption, thus suggesting a need for fuel-efficient vessels.[13,14] Furthermore, end-of-life recycling yields substantial GWP credits, particu-

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larly for metals, although blade recycling remains challenging. These findings underscore the importance of optimizing material use, adopting low-carbon manufacturing (e.g., Electric Arc Furnace for steel), and improving recycling technologies to minimize GWP. Ultimately, while offshore wind offers access to superior energy resources, its higher GWP underscores the critical need for innovation in materials, manufacturing, and circular EoL strategies to fully maximise its potential as a cornerstone of a sustainable energy future.

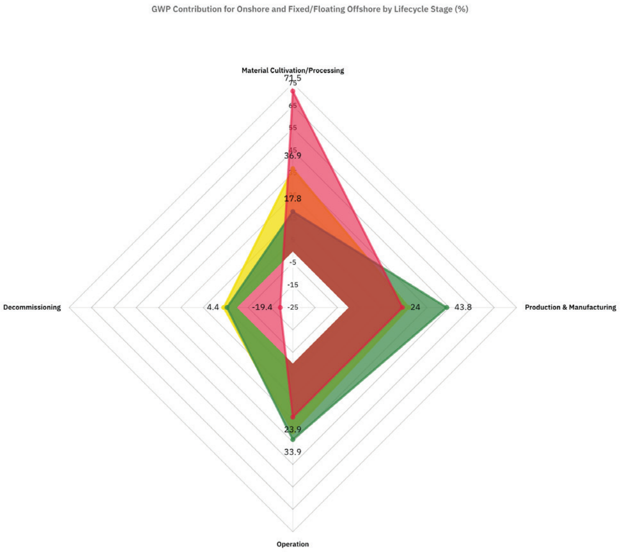

Figure 2: GWP contribution for onshore and fixed/floating offshore wind by life-cycle stage (%). This radar chart follows the same format as Figure 1, comparing the GWP breakdown for onshore (red), fixed offshore (green), and floating offshore (yellow) wind turbines across four life-cycle stages. The comparison reveals that while onshore wind's footprint is concentrated in material processing, fixed and floating offshore systems exhibit more distributed impacts with significantly higher relative burdens from production and operations. This suggests how moving wind energy to marine environments shifts the environmental impact from raw material intensity toward complex infrastructure manufacturing and vessel-based maintenance. Data synthesized from Nugent & Sovacool (2014),[8] Lotfizadeh (2024),[15] and the Vestas LCA reports (Garrett & Mali, 2022; Allekotte & Garrett, 2024).[13,14]

Life-Cycle Analysis of Microalgae Biofuel: Overview and Scope:

Microalgae biofuel production is a biological and chemical process that converts photosynthetic microorganisms into usable fuels. Unlike the discrete hardware of solar and wind, its system is a multi-stage process that is highly dependent on and influenced by the chosen pathway.[17]

It first begins with cultivation, where microalgae are grown in either large, open raceway ponds or in enclosed, controlled photobioreactors (PBRs).[17] During this stage, the algae absorb CO_2 and nutrients to create biomass.[17] Next, this biomass must be harvested (which is an energy-intensive dewatering step), and its valuable compounds, like oils, are extracted, often using solvents.[17] Finally, these compounds are converted into fuel through chemical or biological processes, such as transesterification to create biodiesel or anaerobic digestion to create biogas.[17] The LCA for this technology is therefore a measure of the energy and material inputs for each of these distinct stages.

This section examines the lifecycle of microalgae biofuel, a renewable energy source derived from photosynthetic microorganisms. The analysis covers several key conversion pathways, including transesterification for biodiesel and anaerobic digestion (AD) for biogas. A defining characteristic of microalgae biofuel is the extreme variability of its environmental performance, which is highly dependent on the chosen cultivation and conversion technology. GWP values reported in the literature range from being a significant carbon sink in highly optimized systems (as low as -622.9 g CO_2-eq/kWh) to exceptionally high in energy-intensive, unoptimized scenarios (over 4,700g CO_2-eq/kWh).[17,18] This highlights the importance of system design in determining the technology's sustainability.

Key Terminology and Concepts:

To understand the sustainability of these systems, several key concepts must be defined. Eutrophication refers to nutrient overload that causes water pollution, while acidification contributes to the formation of acid rain. Transesterification is a key chemical reaction in biodiesel production, while common conversion technologies include Anaerobic Digestion (AD), which produces biogas, and Hydrothermal Liquefaction (HTL), which creates bio-oil. Key performance metrics include the Net Energy Ratio (NER), which measures energy output relative to input; the High Heating Value (HHV), which quantifies a fuel's energy density; and the C:N:P Ratio, which is crucial for optimizing microalgae growth. Finally, some studies aggregate various environmental impacts into a single score using the unit mPt (millipoints), where one point (1000 mPt) represents one-thousandth of the annual environmental load of an average European inhabitant.

Material Cultivation and Processing:

Microalgae cultivation occurs in open raceway ponds or closed photobioreactors (PBRs). Open ponds are cost-effective and scalable but prone to contamination and require significant land area, though they can be scaled efficiently once land availability is met. PBRs use transparent tubes or panels, often illuminated by sunlight with mirrors or LED lamps to enhance photosynthesis, at the cost of higher energy inputs (5–10 kWh/m^3 for lighting and circulation).[17] During cultivation, microalgae absorb CO_2 and assimilate nutrients like nitrogen and phosphorus. A 200-m^3 raceway pond sequesters 358,358 kg CO_2 annually but requires 200.2 kg of swine manure nutrients, which emit 0.5 kg CO_2-eq per kg of manure nutrients during production, yielding a net sequestration of 357,858 kg CO_2 annually.[19]

The conversion of microalgal biomass into energy-dense precursors is an energy-intensive process. For biodiesel production, triacylglycerols (TAGs) are the key intermediate. One study found that producing 1 kg of TAGs from biomass with a 50% TAG content requires 16.18 kWh of energy.[19] For biobutanol production, glucose is the precursor, and producing 1 kg of glucose from biomass with a 50% glucose content requires a much lower 1.22 kWh of energy.[19] To provide context, two large-scale, open-pond cultivation scenarios are examined. A conventional industrial model requires a power demand of

10,808 kW to produce 151,875 kg/hr of biomass.[18] In contrast, an optimized model from the National Renewable Energy Laboratory (NREL) requires a higher power input of 29,057 kW.[18] To contextualize these numbers, the conventional model's power demand is enough to continuously run over 9,000 average U.S. homes, while the optimized NREL model requires enough power for over 24,000 homes. This additional energy supports more complex, integrated processes, such as nutrient recycling and biogas production from waste, which ultimately leads to greater overall system efficiency and a lower net environmental impact. In this NREL model, the cultivation stage contributes approximately 50% of the total lifecycle GWP (108 g CO_2-eq/kWh out of a total 216 g CO_2-eq/kWh), even when using renewable energy.[17]

Production and Manufacturing:

The production phase converts biomass into biofuels via transesterification (biodiesel), fermentation (biobutanol), or pyrolysis. For biodiesel, a process using 10,000 kg/hr TAG, 2,171 kg/hr methanol, and 100 kg/hr sodium hydroxide produces 10,030 kg/hr biodiesel and 1,517 kg/hr glycerol, using a 113,489-kg reactor.[19] Hexane-based extraction requires 14,561.7 kW, processing 2,885,625 kg/hr feedstock (1,442,813 kg/hr recycled), yielding 28,580 kg/hr raw oil, with the lipid upgrading phase needing 1,058 kW to produce 28,713 kg/hr biodiesel.[18] This extraction accounts for 94% of emissions (4,467 g CO_2-eq/kWh) due to hexane use.[18] An optimized system achieves lower emissions by using renewable energy.[18] Lipid extraction (the process of separating the valuable oils (lipids) from the rest of the algal cell matter), in particular, is highly energy-intensive, with some studies reporting that this step alone can consume up to 85% of the total energy required for the entire biofuel production process.[20] Emerging techniques, like wet extraction and supercritical CO_2 extraction, reduce impacts.[20]

For biobutanol, one study describes a simulated process in which 503,377 kg/hr of glucose is fermented in an acetone-butanol-ethanol (ABE) system, producing 51,497 kg/hr of biobutanol with a significant cooling duty of 335 MW.[19] Biomass productivity and energy sources significantly affect emissions; for example, switching the source of electricity from hard coal to hydropower can reduce the GWP of electricity generation by over 99%.[3]

The energy efficiency of different conversion pathways varies significantly. The NER is a key metric, and for the pathways analyzed, NER values are reported as 1.524 for transesterification, 1.248 for HTL, and 2.463 for pyrolysis.[17] An NER greater than 1 indicates a net energy loss, making pyrolysis particularly energy-intensive. This is demonstrated in a lab-scale study of *Desmodesmus* sp. EJ 8-10, where the harvest stage alone consumed 56.24% of the total lifecycle energy, resulting in a high calculated GWP of approximately 498.8 g CO_2-eq/kWh for the entire process.[21] In contrast, anaerobic digestion (AD) can be remarkably energy-positive. For a conceptual 100-hectare facility producing biogas from *Chlorella vulgaris*, the total daily electrical input is 16,000 kWh.[22] The gross energy generated from the produced biogas is approximately 65,300 kWh per day. After accounting for the system's internal energy needs for heat (17,000 kWh/day) and electricity (2,694 kWh/day), the facility still demonstrates a strongly positive energy balance. Hence, this shows that a well-designed AD system can generate far more energy than it consumes in its daily operation.[22]

Operational Phase:

The operational phase involves combustion, releasing emissions over the lifecycle. A cradle-to-gate boundary was used in some studies, while others adopt a well-to-wheel scope.[18,19] Lifecycle emissions for unoptimized systems can be extremely high; for example, one conventional pathway reports a net GWP of 4,752 g CO_2-eq/kWh, a value that remains high even after accounting for the CO_2 uptake during cultivation due to massive process emissions from solvent use.[18] This value can be dramatically reduced through system optimization and the use of co-product credits. Co-product credits are an LCA accounting method that subtracts the emissions that would have been generated by conventionally producing the co-products. For example, in an optimized system, credits from valorizing co-products like ethanol and biogas can lower the final net GWP to 159 g CO_2-eq/kWh.[18] While process improvements over time are evident, the specific GWP values depend heavily on the system design and assumptions used.

In a system producing biogas via anaerobic digestion, the combustion of algal methane is a dominant source of emissions, contributing approximately 338.4 g CO_2-eq/kWh.[22] This large emission is then significantly counterbalanced by the CO_2 sequestered during the algae's growth phase, which greatly reduces the final net GWP of the entire system. Other emissions include photochemical oxidation (32.9% of total impact per MJ) and minor contributions to eutrophication (22.8%) and acidification (11.5%), reflecting exhaust byproducts like NOx and SOx.[22] When compared on a life-cycle basis to petroleum diesel, the high electricity demand for the algal methane process can increase its GWP. This impact can be partially mitigated in systems that recycle CO_2 from biogas upgrading back into the cultivation ponds, where the microalgae can capture up to 90% of the injected CO_2.[22]

Net-negative GWP values are possible in highly optimized systems where the CO_2 sequestered during cultivation, combined with credits from the valorization of co-products, exceeds the total lifecycle emissions. For example, one study reports GWP values as approximately -37.3 g CO_2-eq/kWh for transesterification, -24.4 g CO_2-eq/kWh for HTL, -622.9 g CO_2-eq/kWh for AD without pretreatment, and -219 g CO_2-eq/kWh for AD with pretreatment.17 Another study reports GWP values of -48.09 g CO_2-eq/kWh for AD without pretreatment and -16.91 g CO_2-eq/kWh for AD with pretreatment.[17] On the other hand, pyrolysis has a high positive GWP of approximately 954.9 g CO_2-eq/kWh, as it lacks significant co-product credits.[17] The potential for net-negative emissions in other pathways is driven by CO_2 fixation during cultivation, which can be as high as -125.68 g CO_2-eq/kWh for AD without pretreatment.[17] In contrast, a lab-scale total GWP of approximately 498.8 g CO_2-eq/kWh for *Desmodesmus* sp. EJ 8-10 highlights the impact of unoptimized, energy-intensive

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processes.[21] Other impacts from this process include acidification (0.86 g SO_2-eq/kWh), eutrophication (0.065 g PO_4-eq/kWh), and photochemical ozone synthesis (0.047 g C_2H_4-eq/kWh).[21]

End-of-Life Management:

End-of-Life (EoL) management for microalgae biofuel differs from that of durable energy systems. Because the fuel's end-of-life is its combustion (accounted for in the operational phase), the EoL stage for biofuel production focuses on valorizing process residues to improve overall sustainability and create a more circular system. This strategy can significantly offset the initial environmental impacts of production. For example, co-products such as glycerol (1,517 kg/hr) and residual biomass can be used to produce biogas via AD, providing both electricity and digestate as fertilizer.[19,23] Stillage recycling reduces emissions, while discarding 1,933,968 MJ/hr of residue increases impacts.[17]

AD systems benefit from nutrient recycling (70% of total consumption) via the residual liquid phase, reducing fertilizer needs, while transesterification, HTL, and pyrolysis lack this due to toxic by-products (e.g., nitrogen heterocycles, fatty acids, nickel).[17] Anaerobic digestion of Chlorella vulgaris produces 9,385 m^3/day of biogas (70% CH_4) from a 100-ha facility, with 30% (2,816 m^3/day) burned to heat digesters (17,000 kWh/day) and 70% (6,570 m^3/day) upgraded to 5,061.5 m^3/day of 96% CH_4 fuel.[22] The gross energy generated from the produced biogas is approximately 65,300 kWh per day, which, after accounting for the system's internal energy needs for heat and electricity, demonstrates a strongly positive energy balance.[22] Nutrient recycling is significant: 90% of nitrogen (N), phosphorus (P), and potassium (K) in digestates is mineralized, reducing fertilizer needs by an equivalent amount (e.g., 61 gN, 8.1 gP, 6.59 gK per kg dry algae).[22] However, solid digestates used as soil conditioners may contribute to eutrophication if not managed properly.[22]

Synthesis and Implications:

The GWP of microalgae biofuel is not a single value but rather a wide spectrum, highly contingent on the chosen conversion pathway. As visualized in Figure 3, the environmental footprint differs dramatically between conventional and optimized systems. For optimized systems, particularly those using AD with co-product and nutrient recycling, the GWP can be net-negative, with values reported as low as -622.9 g CO_2-eq/kWh.[17] At the other extreme, energy-intensive processes like the lab-scale pyrolysis of Desmodesmus sp. EJ 8-10 can result in high emissions, calculated at approximately 498.8 g CO_2-eq/kWh.[21] This vast range indicates that while the most optimized microalgae systems can outperform wind (34.1 g CO_2-eq/kWh) and solar PV (49.9 g CO_2-eq/kWh), unoptimized pathways are far less sustainable than established renewables.[8] They can also perform worse than many first-generation biofuels; for context, the average GWP for corn starch ethanol is 277.2 g CO_2-eq/kWh and for soybean biodiesel is 156.6 g CO_2-eq/kWh.4 While these first-generation fuels also have wide performance ranges, the high-end emissions from some microalgae pathways far exceed them.

The benefits of integrated systems are also clear when compared to fossil-based production. One study found that a microalgae-to-butanol chain had an overall environmental impact score (162 mPt) that was 62.7% lower than a conventional crude-oil-to-butanol process (466 mPt).[19] Furthermore, valorizing co-products through processes like anaerobic digestion can provide significant energy credits, further lowering the net GWP.

Scalability remains an obstacle due to high energy inputs, a key reason microalgae biofuel has not been widely adopted.[22] Despite environmental benefits like CO_2 sequestration, microalgae biofuel struggles to compete with other renewables or fossil fuels. Its practical use is limited by the lack of reasonably scalable cultivation methods. These methods currently involve high costs, significant land use, and large energy demands.[22] Crucially, the sustainability of these energy-intensive processes is almost entirely dependent on the source of electricity. As highlighted earlier, switching from a fossil-fuel-based grid to a clean source like hydropower can reduce the GWP of the required electricity by over 99%.[3] Therefore, integrating renewable energy is not just an improvement, but a fundamental requirement for making microalgae biofuel a viable, low-carbon alternative.

Integrating renewable energy and carbon capture and storage (CCS) can further reduce GWP. CCS captures 60–80% of CO_2 emissions during cultivation and processing, such as those from electricity generation or combustion of by-products in pyrolysis and AD systems.[22] For microalgae, CCS can be applied at the cultivation stage by capturing CO_2 from nearby industrial sources (e.g., power plants) and supplying it to raceway ponds, enhancing growth while reducing net emissions.[22] During processing, CCS can capture emissions from energy-intensive steps like drying or HTL, where fossil energy is often used, potentially lowering GWP by an additional 50–100 g CO_2-eq/kWh in optimized systems.[18,22] This integration is highly compatible with microalgae's natural CO_2 sequestration ability, thereby providing a potential co-benefit for the technology's overall sustainability. Research must prioritize energy-efficient extraction, scalable cultivation, and co-product valorization to overcome these hurdles.

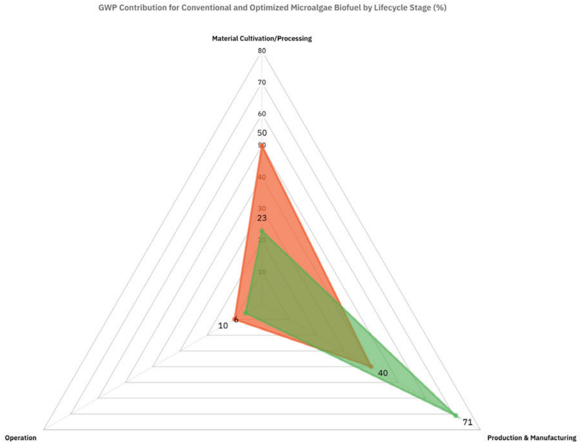

Figure 3: GWP contribution for conventional and optimized microalgae biofuel by life-cycle stage (%). This triangular plot compares the GWP breakdown for a conventional (green) and an optimized (orange) microalgae system. A decommissioning phase is not applicable as the fuel's end-of-life is its combustion, which is included in the operational phase. The comparison indicates that while conventional methods are burdened by energy-intensive production and manufacturing, optimized systems utilize biological sequestration and nutrient recycling to transition the technology into a potential net carbon sink. This emphasizes how the environmental viability of microalgae biofuel is fundamentally determined by system design and the level of integration with circular economy principles. Data synthesized from Dutta *et al*. (2016),18 and Sun *et al*. (2019).[17]

Comparison and Summary:

The life-cycle analysis of solar photovoltaic (PV) systems, wind energy, and microalgae biofuel reveals that no single technology is universally superior. Instead, their environmental performance is highly dependent on technological choices, geographical context, and operational scale. While modern onshore wind turbines often serve as a benchmark for low greenhouse gas (GHG) emissions, studying them closer reveals significant variations across all technologies depending on specific conditions and system designs. These factors are summarized in **Table 1**, which compares the lifecycle GWP and EPBT for key renewable technologies.

Table 1: Comparative life-cycle GHG Emissions of renewable energy technologies.

Technology	**Specific Type / Scenario**	**Total Lifecycle GWP (g CO_2-eq/kWh)**	**EPBT (years)**
Solar PV	Crystalline Silicon (1st Gen)	**~49.9**	**2.2 – 6.0**
	Cadmium Telluride (CdTe) Thin-Film (2nd Gen)	**19 – 30**	**0.8 – 3.6**
	Quantum Dot (QD) PV & other 3rd Gen	**2.89 – 150**	**0.24 – 1.2**
Wind Energy	Onshore (Modern Turbine, e.g., Vestas V136)	**5.6 – 36.78**	**< 1**
	Offshore (Fixed Foundation, e.g., Vestas V236)	**7.0 – 26.15**	**< 1**
	Offshore (Floating Foundation, e.g., Hywind)	**~36.78**	**< 1**
Microalgae Biofuel	Anaerobic Digestion (Optimized)	**-48 – -623**	**N/A**
	Transesterification (Optimized)	**~ -37**	**N/A**
	Pyrolysis (Unoptimized)	**~955**	**N/A**
	Conventional/Unoptimized Pathways	**>4,000**	**N/A**

Note: Negative GWP values for microalgae reflect systems where CO_2 uptake and avoided emissions from co-products exceed total process emissions. Single GWP values (e.g. ~49.9) typically represent a mean from a meta-analysis or a result from a specific case study, whereas ranges capture the variability across multiple studies. The EPBT for wind energy is consistently under one year across most studies; EPBT is not applicable (N/A) for microalgae biofuel, as the metric is used for direct energy-generating systems, not liquid fuel production. This table highlights that modern onshore wind turbines currently offer the most consistent and lowest lifecycle emissions among the evaluated technologies. Conversely, microalgae biofuel presents the widest performance range, which demonstrates that its environmental viability ranges from a net carbon sink to a high emitter depending on system design and technological maturity. Data for Crystalline Silicon PV sourced from Nugent & Sovacool (2014).[8] Data for CdTe PV sourced from Peng *et al*. (2013).[5] Information for 3rd Gen PV comes from Muteri *et al*. (2020).[6] Data for Wind Energy sourced from Garrett & Mali (2022),[13] Allekotte & Garrett (2024),[14] Lotfizadeh (2024),15 and Bonou *et al*. (2016).[9] Data for Microalgae Biofuel sourced from Sun *et al*. (2019),[17] and Dutta *et al*. (2016).[18]

In particular, onshore wind energy emerges as a leading technology due to its low material intensity and high recyclability, resulting in a rapid EPBT of less than a year. However, its performance is geographically constrained and faces challenges related to land use and social acceptance. Offshore wind, while accessing stronger and more consistent wind resources, carries a higher environmental footprint due to the increased material requirements for foundations and more complex maintenance logistics.

Solar PV technology, particularly crystalline silicon, is hindered by the energy-intensive nature of silicon purification. Its competitiveness is therefore strongly tied to the carbon intensity of the manufacturing grid. Thin-film technologies like CdTe and emerging options like Quantum Dot PV offer lower-emission alternatives but face their own challenges regarding material toxicity and commercial scalability.

Microalgae biofuel presents the most extreme performance variability. In optimized, integrated systems that valorize co-products and recycle nutrients, it can act as a net carbon sink. However, conventional, energy-intensive cultivation and harvesting methods render it far less sustainable than wind or solar. Its scalability is also severely constrained by high water and nutrient demands, making it a high-risk, high-reward technology. This profile makes it best suited for niche applications, particularly as a "drop-in" fuel for sectors like aviation and long-haul freight, which are difficult to electrify with current battery technology.[24] In these areas, biofuels represent one of the only viable pathways to decarbonization in the near future.[24]

Ultimately, the most sustainable renewable energy source depends on context, not on a universal ranking. This review's central finding is that the optimal choice is determined by a combination of regional climate, technological maturity, and logistical factors. For instance, climatic conditions make solar PV the logical choice for arid, high-insolation regions like the Sahara. In contrast, the consistently strong winds of the North Sea make it the ideal location for offshore wind development, which is why countries in that region are heavily investing there. Logistical factors are also crucial, because the life-cycle sustainability of solar PV is maximised when panels are manufactured using a low-carbon electricity grid. This is demonstrated by manufacturing in regions that utilize hydropower, such as in Scandinavia, or geothermal energy, as seen in Iceland.[25,26] Similarly, microalgae biofuel becomes most viable when co-located with industrial facilities that can provide a steady supply of waste CO_2 and nutrients, transforming a logistical challenge into a mutually beneficial situation. In contrast, the lower maturity of microalgae biofuel means it is better suited for regions with strong R&D infrastructure. The optimized NREL model in the United States is a good example, as its viability relies on advanced R&D and a policy focus on developing circular bio-economics.[18] This all supports the thesis that where a technology is deployed and manufactured is as important and worth taking into consideration as the technology itself.

Conclusion

This comparative LCA demonstrates that onshore wind energy currently offers the lowest and most consistent greenhouse gas emissions. However, the analysis also shows that no single renewable technology provides a universally optimal solution for decarbonization. The sustainability of solar PV, wind, and microalgae biofuel is determined by regional context, technological maturity, and the integration of circular economy principles. These findings support the argument that the path to a sustainable energy future requires an approach customized to specific geographic and industrial conditions, rather than reliance on a single technology.

To accelerate this transition, a coordinated effort is indispensable. Policymakers should implement region-specific incentives that favor technologies aligned with local resources, such as prioritizing solar PV in high-insolation areas and supporting wind projects where winds are consistent. Furthermore, policies must mandate and support the development of robust recycling infrastructure for PV panels and turbine components to minimize end-of-life impacts. In parallel, industry must invest in decarbonizing manufacturing supply chains, particularly for solar PV. Industry must also advance the commercial-scale deployment of next-generation technologies. For microalgae, this means focusing on integrated systems that utilize industrial waste streams and employ energy-efficient processes like anaerobic digestion. Finally, researchers have a critical role in closing remaining data gaps in LCAs, including the long-term performance of offshore wind farms, the real-world efficiency of advanced PV recycling, and the scalability of optimized microalgae systems. By aligning policy, investment, and research, the environmental performance of all three technologies can be significantly enhanced, paving the way for a resilient and sustainable global energy system.

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I attest that the ideas, graphics, and writing in this paper are entirely my own.

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