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Systematic Review

A Systematic Review on Environmental Life Cycle Assessment of Solar PV Modules

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Abstract

This review examines an environmental LCA of solar modules and cell technologies across 18 environmental indicators to assess the performance of current solar module types. It provides a more comprehensive analysis by including studies that account for recycling credits in end-of-life PV waste management. The literature search covered seven databases up to 20 November 2025, resulting in the selection of 43 papers focused on solar modules, LCA, and recycling for data extraction. The methodological quality and risk of bias of included studies were evaluated based on compliance with ISO 14040 and ISO 14044 requirements. Due to the diversity of methodologies, functional units, and system boundaries across studies, it is observed that there is a wide range of emission values for the various solar PV technologies, with some studies reporting very high or very low figures. Considering one of the main impact categories, GHG emissions of key PV technologies, including mono c-Si, multi c-Si, CdTe, and PERC modules, for the study period 2015–2025 have been reported in the ranges of 493–2760, 640–2418, 312–2140, and 425–1759 kg CO_2-eq/kW_p. It can be generally expected that when assessed under comparable LCA frameworks, emerging solar cells are more likely to exhibit lower emissions than conventional silicon-based solar cells across most indicators. Moreover, results from recent studies show an environmental improvement for the various module and cell types, largely due to advancements in material efficiency during the optimized manufacturing process. A major limitation of this study is the omission of service lifetime considerations, as most sustainability assessments are ideally based on lifetime energy generation (kWh). However, given the assumption of comparable lifetimes among the assessed module types, the use of kW_p-based emissions remains suitable for relative comparisons, although possible differences in long-term performance may not be fully captured.

Keywords: solar photovoltaics; life cycle assessment; solar modules recycling; climate change

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1. Introduction

Global cumulative PV capacity reached 2.2 TW at the end of 2024 [1], up from 1.6 TW in 2023 [2], and it is projected to increase further to 5.1 TW by 2028 [3]. China alone contributed over 1 TW of cumulative capacity, representing nearly 45% of the global installed capacity, and its PV industry is well-positioned for continued growth [1]. In 2024, solar modules and their components primarily came from China, accounting for 80% of the global module production and more than 95% of the market for upstream components, such as ingots and

wafers [4]. Chinese solar manufacturers top the sector for 2024, showing China's sustained dominance in large-scale solar module manufacturing [5].

The rest of the world represents the remaining 55% of the global cumulative PV capacity. Within this share, the EU and Germany accounted for approximately 340 GW and 100 GW of installed capacity, respectively, in 2024. Along with China and Germany, the USA, India, and Japan completed the list of the world's top 5 PV markets with approximately 224 GW, 125 GW, and 97 GW, respectively [1]. With the increase in solar installation capacity, it is expected that the demand for resources associated with solar technologies will also grow. These resources encompass not only the energy required for manufacturing solar technologies but also the materials used, for which the availability, scarcity, potential environmental, or health aspects must be considered. In the context of climate change and the ever-increasing material constraints, it becomes essential to develop innovative solar technologies that are more resource-efficient and have lower environmental burdens. Moreover, these technologies need to be developed under the principle of eco-design, which integrates the environmental aspects into the product development process while maintaining a balance between ecological performance and economic considerations.

By 2030, cumulative global PV waste is estimated to reach 1.7–8 million tonnes (4–14% of total installed capacity) and 60–78 million tonnes (approximately 80% of total installed capacity) in 2050 based on an average PV panel lifespan of approximately 25 years [6]. It is estimated that about 33,205 tonnes of PV cell waste can generate 62.26 tonnes of hazardous materials; therefore, 78 million tonnes in 2050 could generate as much as 150,000 tonnes of hazardous waste [7]. Therefore, the eco-design of these innovative solar technologies should be considered under the life cycle thinking perspective, in other words, minimizing their negative environmental impacts in all stages of the technologies' life cycle by identifying, quantifying, and assessing the material and energy flows associated with PV production, use, and end-of-life/waste management processes [8].

Several review studies have previously covered topics related to LCA, solar PV modules, and end-of-life PV recycling, but according to the research done by the authors, only seven such reviews have been published since 2015. In 2016, Chatzisideris et al. [9] conducted a comprehensive review on the LCA of thin film PV technologies from an eco-design perspective, examining 33 LCA studies. Their work emphasized the need for holistic assessments, including the eco-design aspects, of which only five studies covered the entire life cycle and impact categories, highlighting the importance of going beyond limited indicators like climate change or energy use to avoid burden shifting. Subsequently, Ludin et al. [10] analyzed LCA studies across various solar PV technologies, including silicon solar cells, thin-film solar cells, dye-sensitized solar cells (DSSCs), perovskite solar cells (PSCs), and quantum dot-sensitized solar cells (QDSSCs), focusing on cumulative energy demand (CED), energy payback time (EPBT), and GHG emissions. Their findings support those of Chatzisideris et al. [9], indicating that prior to 2018, most reviews primarily concentrated on two energy- and climate-related indicators. The updated review by Muteri et al. [11] on the existing LCA literature covering first- to third-generation PV technologies provided information on energy and environmental metrics. Similarly, Gressler et al. [12] compared different emerging PV systems, such as organic solar cells (OSCs), DSSCs, PSCs, and QDSSCs, in terms of material, energy, and chemical requirements, as well as the environmental hotspots reported in the reviewed LCAs.

LCA studies on the life cycle impacts of both commercially available thin film solar cells (a-Si, CIGS, CIS, CdTe, GaAs, and GaAs tandem), as well as emerging ones (PSC, PSC tandem, DSSC, OPV, CZTS, and QD), were compared in a 2023 study by Maalouf et al. [13]. Their analysis also included a comparison with silicon-based solar cells, focusing on indicators such as CED, EPBT, and GHG emissions, and a brief overview of other environmental

impacts. In terms of recycling, Mao et al. [14] provide an overview of recycling end-of-life PV panels, comparing the impacts of landfilling, downcycling, and upcycling across six impact categories (climate change, terrestrial acidification, human toxicity, fossil resource scarcity, water eutrophication, and marine eutrophication) but without considering the full life cycle impacts associated with PV manufacturing. Meanwhile, Cellura et al. [15] presented an LCA review on the emerging solar technologies considering four environmental indicators: energy consumption, GHG emissions, material depletion, and toxicity. Although that study aligns closely with the objectives of this review, it did not fully address the influence of recycling credits on environmental performance. The review was also specific to energy-related and material-based indicators, overlooking several other relevant impact categories.

The objective of this paper is to provide a comprehensive review of recent research on life cycle assessment of current PV module designs incorporating recycling credits across a range of environmental impact categories, i.e., climate change, acidification, eutrophication, toxicity, etc. This paper identifies the current research trends in solar technologies and reports on all key environmental impacts during the life cycle of the current solar PV technologies and the emerging new module designs. All this information is intended to support the eco-design of the new improved "designed for recycling" solar technologies, offering valuable guidance not only for energy and environment engineers but also for policy makers, seeking to promote the development of higher-efficiency solar cells with improved environmental performance.

2. Materials and Methods

This study presents a comprehensive review of LCA studies on solar modules, carried out through a process that included formulating the research questions, identifying relevant literature, assessing literature quality, extracting data, synthesizing findings, and interpreting the results. This study followed a quality assessment approach of LCA studies conducted based on the International Organization for Standardization for life cycle assessment (ISO 14040 and ISO 14044) [16,17]. The review followed the updated version of PRIMA guidelines [18]. This review protocol was not registered in a public registry, as no dedicated registry is available for life cycle assessment studies. However, the review methodology was defined in advance and followed established life cycle assessment standards. The LCA methodology is currently standardized upon the two universal standards ISO 14040 and ISO 14044. ISO 14040: 'Environmental management—Life cycle assessment—Principles and framework' [16] and ISO 14044 (2006b): 'Environmental management—Life cycle assessment—Requirements and guidelines' [17]. They are used as benchmarks to evaluate methodological quality and potential risk of bias in this study by identifying different aspects of the reviewed studies, such as system boundaries, functional units, and impact categories. As represented in Figure 1, this review process also followed the guidelines of the Preferred Reporting Items for Systematic Reviews and Meta-Analyses (PRISMA) 2020 for the identification, screening, and selection of the relevant articles, as well as the statistical analysis of the results [18]. However, in this study, PRISMA 2020 was applied as a framework to enhance transparency rather than being followed as a rigid procedure. The Supplementary Information provides an overview matrix of the PRISMA used. For analyzing methodological aspects, such as system boundaries, functional units, and impact categories, in the selected LCA studies, the review followed an established systematic literature review framework tailored to life cycle thinking research. In the first stage, the research questions of this review paper were formulated after an initial assessment of the existing literature on the same topics of LCA and solar modules, resulting in the identification of three key research questions:

- How do existing PV module designs perform in terms of sustainability and GHG emissions, including recyclability, when assessed using various recycling technologies?
- What are the most material- and energy-intensive stages/processes in the life cycle of solar modules?
- What are the environmental impacts of new, improved module designs for recycling strategies and the associated end-of-life recycling?

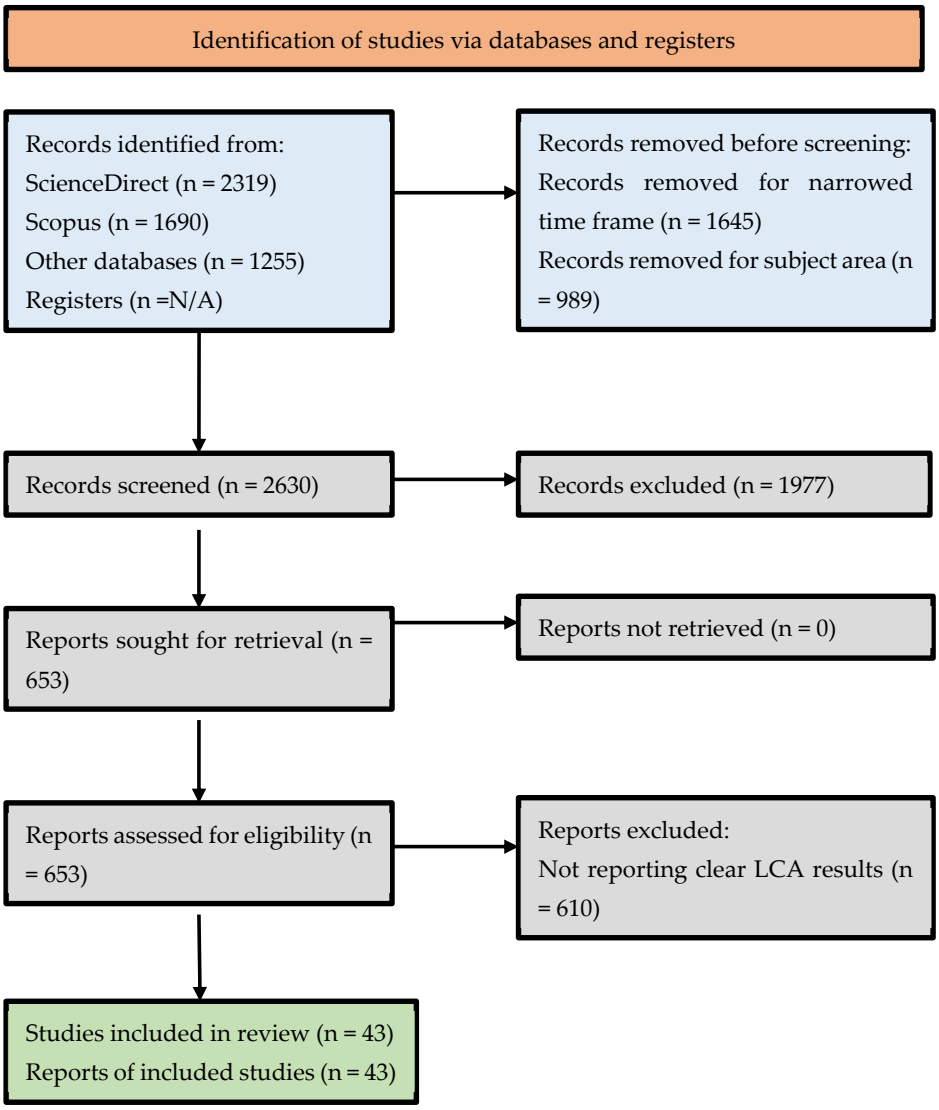


Figure 1. Systematic literature review flow diagram adopted from PRISMA.

In the second stage, the keywords and timeframe for identifying relevant studies were defined. Two keywords related to the research topic were used—"life cycle assessment" and "solar photovoltaics"—targeting research focused on the material scale of solar systems and the LCA methodology. The search string "solar photovoltaics" AND "life cycle assessment" was applied across both Science Direct and Scopus databases. The search also utilized additional thematic keyword combinations, such as "Recycling of PV panels", "Environmental impact of recycling PV panels", and "Sustainable recycling process of PV waste", for the primary search on recycling in the above-mentioned databases, as well as Google Scholar, Web of Science, SpringerLink, Wiley Online Library, and IEEE Xplore. In total, the search process yielded 5264 studies published between 2000 and 2025.

In the secondary search phase, a systematic filtering process was applied to identify the most relevant studies. The publication period was narrowed down to between 2015 and

2025 to capture recent developments in LCA methodologies and PV technologies, including improved efficiency, use of innovative materials, etc. The retrieved studies were further filtered and screened based on the document type (research and review articles) and subject area (Energy, Environmental science, Materials science, and Engineering), reducing the total number of papers to 2630. Subsequent title screening focused on studies addressing existing research trends, such as countries studied, module/cell types, impact categories, and lifecycle stages, followed by abstract screening to ensure inclusion of core LCA concepts. The selection criteria involved a careful evaluation of each paper's content, title, abstract, and keywords, requiring that studies be peer-reviewed within the past decade. After the complete skimming and screening process, 43 papers focusing on solar modules, LCA, and recycling were selected for data extraction. The extracted data on environmental impacts and recycling of current PV module designs were then analyzed in Excel spreadsheets and synthesized for interpretation.

The assessment of the quality of the selected papers for data extraction was based on six key methodological aspects, including system boundaries, functional units, inventory data, the LCIA method, impact categories, and the inclusion of sensitivity analyses. These criteria were evaluated across the four main phases of Life Cycle Assessment, i.e., Goal and scope definition, Life Cycle Inventory Analysis (LCI), Life Cycle Impact Assessment (LCIA), and Interpretation, in accordance with ISO 14040 series. Each study was assigned one of three risk-of-bias ratings: low risk, moderate risk, and high risk. Studies were considered high risk if one or more of the assessed criteria were rated as high risk. Those with moderate risk have at least two criteria classified as moderate risk and none as high risk, while studies classified as low risk have no high-risk ratings and only one criterion rated as moderate risk. Table 1 shows the main studies used for this review with a legend for module/cell type and impact categories in Table 2.

Table 1. The main studies used for the review.

Study	Study Countries	Module/Cell Type	Life Cycle Stages	System Boundaries	Functional Unit (FU)	Impact Categories
[19]	China	Multi c-Si	Manufacturing	Cradle to gate	1 kW_p	a, b, e, f, h, i, j, k, l, m
[20]	Italy	Solar PV waste	End of Life (Recycling)	Gate to gate/grave	1000 kg	a, b, c, d, e, f, h, i, l, m, n, o, r
[21]	EU	Mono c-Si, Multi c-Si, Micro c-Si, CdTe, CIGS	Manufacturing, Installation, Use, End of Life	Cradle to grave	1 kWh	a, b, c, d, e, f, h, i, l, m, n, o, p, q
[22]	EU, US, China, Malaysia, Japan	Mono c-Si, Multi c-Si, CdTe, CIGS	Manufacturing, Installation, Use	Cradle to gate (use)	1 kW_p	a, e, m
[23]	Singapore	PERC, Al-BSF	Manufacturing, Installation, Use	Cradle to gate (use)	1 kWh	a
[24]	China, Germany, UK, Spain	Mono c-Si, Multi c-Si	Manufacturing, Installation, Use	Cradle to gate (use)	1 kWh	a, b, e, f, h, i, j, k, l, m
[25]	China	PERC, Al-BSF	Manufacturing, Installation	Cradle to grave	1 kWh	a, b, h, i
[26]	UK	Perovskites (PSC/PST)	Manufacturing	Cradle to gate	1 kWh	a, b, e, f, h, i, l, m
[10]	Global	Mono c-Si, Multi c-Si, a-Si, CdTe, CIGS	N/A	N/A	1 kWh	a
[27]	US	PERC, Tandem	Manufacturing, Installation	Cradle to grave	1 kWh	a, b, h
[28]	Thailand	Solar PV waste	End of Life (Recycling)	Gate to gate/grave	1000 kg	a, b, c, e, f, g, h, i, j, k, l, m, n, o, p, q
[29]	UK, Spain	Mono c-Si, Multi c-Si, a-Si, CdTe, CIGS	Manufacturing, Installation, Use	Cradle to gate (use)	1 kWh	a, b, e, h
[30]	EU	Solar PV waste	End of Life (Recycling)	Gate to gate/grave	1000 kg	a, b, c, d, e, f, h, i, l, m, n, o

Table 1. *Cont.*

Study	Study Countries	Module/Cell Type	Life Cycle Stages	System Boundaries	Functional Unit (FU)	Impact Categories
[31]	Australia	Solar PV waste	End of Life (Recycling)	Cradle to grave and Gate to gate/grave	20 kt	a, b, c, e, f, g, h, i, j, k, m, n, o, p, q
[32]	Global	Multi c-Si	Manufacturing, Installation, Use, End of Life	Cradle to grave	1 kWh	a, b, e, n
[33]	Mexico	Mono c-Si, Multi c-Si, a-Si, CIGS	Manufacturing, Installation, Use, End of Life	Cradle to grave	1 kWh	a, b, e, f, h, i, j, k, l, m
[34]	EU	Solar PV waste	End of Life (Recycling)	Gate to gate/grave	1000 kg	a, b, c, e, f, g, h, i, j, k, l, m, n, o
[35]	China, EU, Germany	PERC	Manufacturing, End of Life	Cradle to grave	1 kW_p and 1 kWh	a, b, c, d, e, f, h, i, l, m, n, o, p, q, r
[36]	China	PERC	Manufacturing, Installation, Use, End of Life	Cradle to grave	1 kWh	a, b, h, i
[37]	Italy, EU	Solar PV waste	End of Life (Recycling)	Gate to gate/grave	24 t	a, b, c, e, f, g, h, m, n, q
[38]	Malaysia	Multi c-Si	Manufacturing, Installation, Use, End of Life	Cradle to grave	1 kWh	a, b, c, e, f, g, h, i, j, k, m, n, o, p, q
[39]	Spain, China	Mono c-Si, Multi c-Si	Manufacturing, Installation, Use	Cradle to gate (use)	1 kWh	a, e
[40]	China	Mono c-Si, Multi c-Si	Manufacturing	Cradle to gate	1 kW_p and 1 kWh	a, e
[41]	Global	Multi c-Si, CdTe, CIGS	Manufacturing, Installation, Use	N/A	1 kWh	a, b, h, p, q
[42]	EU	Mono c-Si, Multi c-Si, CdTe, CIGS	Manufacturing, Installation, Use, End of Life	Cradle to grave	1 kWh	a, e, f, n, q
[43]	Canada	Perovskites (PSC/PST)	Manufacturing, Installation, Use, End of Life	Cradle to grave	1 kWh	a
[44]	EU	Mono c-Si	Manufacturing, Installation, Use, End of Life	Cradle to grave	1 kWh	a, f, i, p
[45]	Canada	Mono c-Si	Manufacturing, Installation, Use	Cradle to gate (use)	1 kWh	a, b, e, h, i
[46]	US, China	Mono c-Si, Multi c-Si, CdTe, CIGS, CZTS, ZN_3P_2, HJT, QDPV, DSSD	Manufacturing, Installation, Use, End of Life	Cradle to grave	1 kWh	a
[47]	China	OSC, PSC	Manufacturing	Cradle to gate	1 m^2	a
[48]	Germany	PST, SHJ	Manufacturing	Cradle to gate	Module and 1 kWh	a, b, c, e, f, g, h, i, j, k, n, o, q
[49]	UK	CIGS, CZTS, CZTSe, CZGeSe, Sb_2Se_3, Sb_2S_3	Manufacturing	Cradle to gate	1 kWh	a, b, e, f, h, i, j, k, l, m
[14]	China	Solar PV waste	End of Life (Recycling)	N/A	N/A	a, b, c, e, f, h
[13]	Global	a-Si, CdTe, CIGS, GaAs tandem, PSC, PSC tandem, DSSC, OPV, CZTS, QD	Manufacturing, Installation, Use, End of Life	Cradle to grave	1 kWh	a
[50]	EU	Mono c-Si, Multi c-Si, CdTe, CIGS	Manufacturing, Installation, Use, End of Life	Cradle to grave	1 kWh	a, e, f, n
[51]	Italy	Solar PV waste	End of Life (Recycling)	Gate to gate/grave	100 kg	a, b, c, e, f, g, h, i, j, k, m, n, o, p, q
[52]	China, Mexico	Solar PV waste	Manufacturing, Installation, Use, End of Life	Cradle to grave	1 kW_p	a, b, c, e, f, g, h, i, j, k, m, n, o, p, q
[53]	US	PERC	Manufacturing, Installation, Use, End of Life	Cradle to grave	1 kWh	a
[54]	Italy	a-Si, CdTe, CIGS, Al-BSF, IBC, PERC, HJT	Manufacturing, Installation, Use, End of Life	Cradle to grave	1 kWh	a, b, c, d, e, f, l, m, n, o

Table 1. *Cont.*

Study	Study Countries	Module/Cell Type	Life Cycle Stages	System Boundaries	Functional Unit (FU)	Impact Categories
[55]	India	Mono c-Si, Multi c-Si, CdTe, CIGS, PERC	Manufacturing	Cradle to gate	1 m^2	a, b, c, e, f, g, h, i, j, k, m, n, o, p, q
[56]	China	Multi c-Si	End of Life (Recycling)	Gate to gate/grave	1000 kg	a, b, c, e, f, g, h, i, j, k, m, n, o, p, q
[57]	Italy	PERC, Al-BSF	Manufacturing, Installation, Use, End of Life	Cradle to grave	1 kWh	a, b, c, d, e, f, l, m, p
[58]	China	TOPCon	Manufacturing	Cradle to gate	1 kW_p	a, b, c, e, f, g, h, i, j, k, m, n, o, p, q

Table 2. Legend for module/cell types and impact categories in Table 1.

Mono c-Si	Mono-crystalline Silicon	a	climate change
Multi c-Si	Multi-crystalline Silicon	b	freshwater eutrophication
Micro c-Si	Micromorphous Silicon	c	marine eutrophication
CdTe	Cadmium Telluride	d	terrestrial eutrophication
CIGS	Copper Indium Gallium Selenide	e	acidification potential
a-Si	Amorphous silicon	f	fossil depletion
PERC	Passivated Emitter and Rear Cell	g	metal depletion
TOPCon	Tunnel Oxide Passivated Contact	h	human toxicity
Al-BSF	Aluminum Back-Surface Field	i	freshwater ecotoxicity
IBC	Interdigitated Bank Contact	j	terrestrial ecotoxicity
HJT	Heterojunction Technology	k	marine ecotoxicity
GaAs	Gallium Arsenide	l	photochemical oxidant formation
PSC	Perovskite solar cells	m	ozone depletion
DSSC	Dye-sensitized solar cells	n	particulate matter formation
OPV	Organic thin-film PV	o	ionizing radiation
CZTS	Copper Zinc Tin Sulfide	p	land use
QD	Quantum Dot	q	water depletion
AZTS	Silver Zinc Tin Sulfide	r	cumulative energy demand
CZTSe	Copper Zinc Tin Selenide		
CZGeSe	Copper Zinc Germanium Selenide		
Sb_2Se_3	Antimony Triselenide		
Sb_2S_3	Antimony Trisulfide		
OSC	Organic solar cells		
PST	Perovskite Silicon Tandem		
SHJ	Silicon Heterojunction		
ZN_3P_2	Zinc Phosphide		

3. Results and Discussion

3.1. Existing Research Trends

A summarized interpretation of the data presented in Table 1 is illustrated in Figure 2, followed by illustrations in Figure 3. Figure 3A shows that among the 43 LCA publications on solar modules completed between 2015 and 2025, the highest number of these (9) took place in 2021. This surge is likely linked to the implementation of the European Green Deal (2019) and the EU Circular Economy Action Plan (2020), which made eco-design, recycling, and sustainability key requirements for energy technologies, prompting increased research in the PV industry to quantify environmental impacts through LCAs. Similarly, the second notable rise in publications on LCAs of solar modules (8) in 2024 coincides with the finalization of the EU Eco-design for Sustainable Products Regulation (ESPR), directly targeting solar PV modules. In relation to the scope of the reviewed studies in Figure 3B, the life cycle stages most frequently evaluated in the LCA studies were classified as (i) manufacturing, (ii) installation, (iii) use/operation, and (iv) end of life/recycling, with transportation considered across all these stages. It shows that manufacturing of PV systems is the most commonly evaluated stage, featuring in ~74% of all LCA studies. Due to

increased eco-design requirements for PV modules, the end-of-life/recycling stage was also widely covered in most studies, especially LCA studies on PV waste, appearing in ~60% of the studies. Installation and use/operation were examined in ~49% and ~44% of the studies, respectively. Regarding the geographical distribution of the studies in Figures 2 and 3C, where the articles on LCA of solar modules are published or where the studied systems are manufactured/installed, China dominates the rankings with a total of 14 papers (~25%), followed by the EU with 9 papers (~16%), Italy with 5 papers (~9%), and the U.S. and the UK with 4 papers each (~7%), forming the top 5 contributors. Germany accounts for 3 published papers (~5%), while other countries such as Spain (3), Canada (2), Mexico (2), Malaysia (2), Japan (1), Singapore (1), Australia (1), India (1), and Thailand (1) each have at least one published paper, with the rest of the world contributing 4 papers (~7%). The domination of China in LCA studies indirectly explains why it is currently dominating the global world market in the manufacturing, development, and implementation of solar PV systems, currently representing 45% of the global installed capacity.

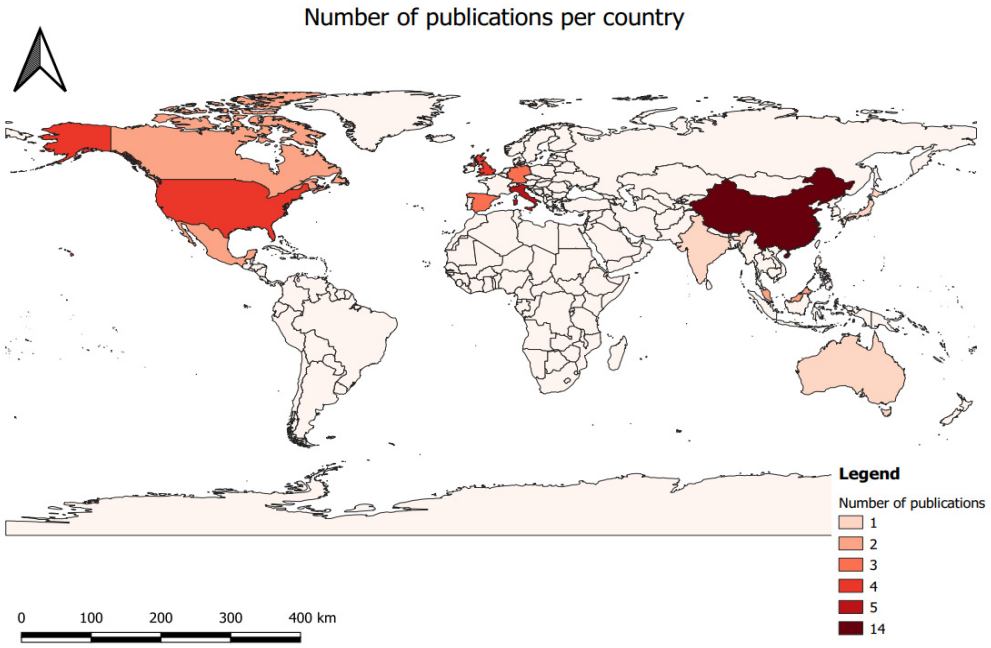


Figure 2. Number of LCA publications on solar modules per country.

In terms of module/cell type in Figure 3D, multi c-Si modules were the most widely studied in these LCA publications (~17%) followed by mono c-Si modules (~14%). However, this distribution is expected to change due to the rising market share of high-efficiency mono c-Si, notably PERC (~9%) and TOPCon (~1%). Thin-film modules, particularly CIGS (~13%) and CdTe (~11%), have also been the focus of several studies. In addition, there are also a number of studies on innovative/emerging solar cells, such as PSC, OSC, and chalcogenide PV cells, many of which remain uncommercialized or in the laboratory phase. In relation to the system boundaries of the reviewed studies in Figure 3E, most studies (44%) utilize the cradle-to-grave approach from manufacturing, installation, use/operation, and end-of-life waste management to understand the environmental impact of the system as a whole. This is followed by the cradle-to-gate and gate-to-gate/grave approaches, both at 18%. The cradle-to-gate approach mainly considers the manufacturing stage, while the gate-to-gate/grave approach only considers the end-of-life waste management through either disposal in landfill (grave) or recycling (gate/cradle). The other approach is the cradle-to-gate (use) approach (14%), which considers the environmental impacts from manufacturing to the use phase without considering end-of-life waste management. Considering the functional units in Figure 3F, kWh is the most utilized functional unit

in 67% of the studies due to the use of the cradle-to-grave approach to studying the system as a whole, while for studies considering only end-of-life stages, most consider the ton (1000 kg) as the functional unit (12%). The other commonly used units are the kW_p and m^2 units. As shown in Figure 3G, climate change (GHG emissions) remains the most frequently evaluated impact category, considered in ~98% of all LCA studies. Other commonly evaluated impacts include acidification potential (~70%), freshwater eutrophication (~67%), and human toxicity (~63%). The dominance of GHG assessments is mainly due to their policy relevance, e.g., the Paris Agreement and EU Green Deal, strong standardization, and ease of interpretation. Finally, while significant progress has been made in documenting key parameters used in LCA studies, the wide variability of these parameters continues to hinder direct comparison between LCA results, as similarly noted by Santoyo-Castelazo et al. [33].

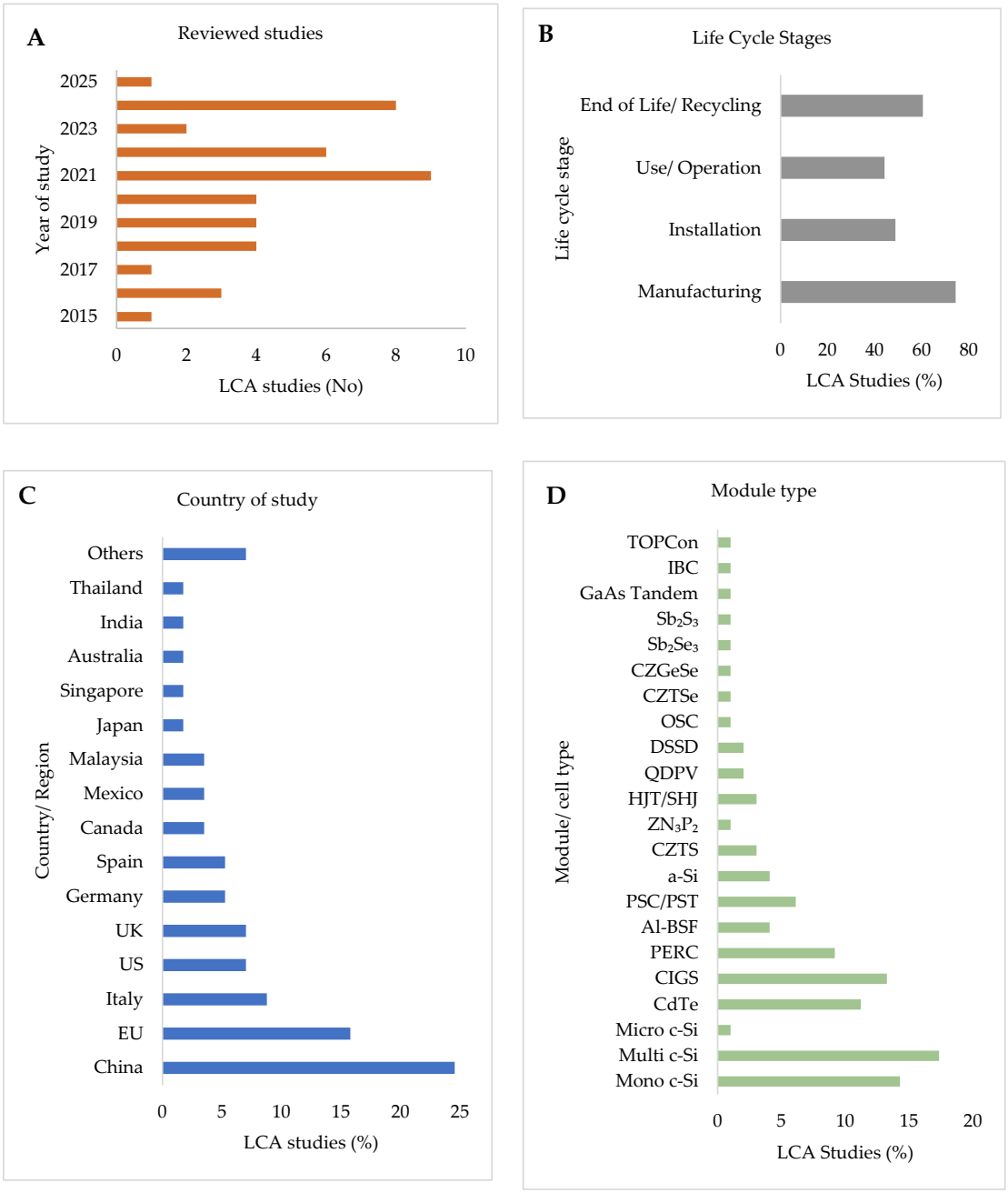


Figure 3. *Cont.*

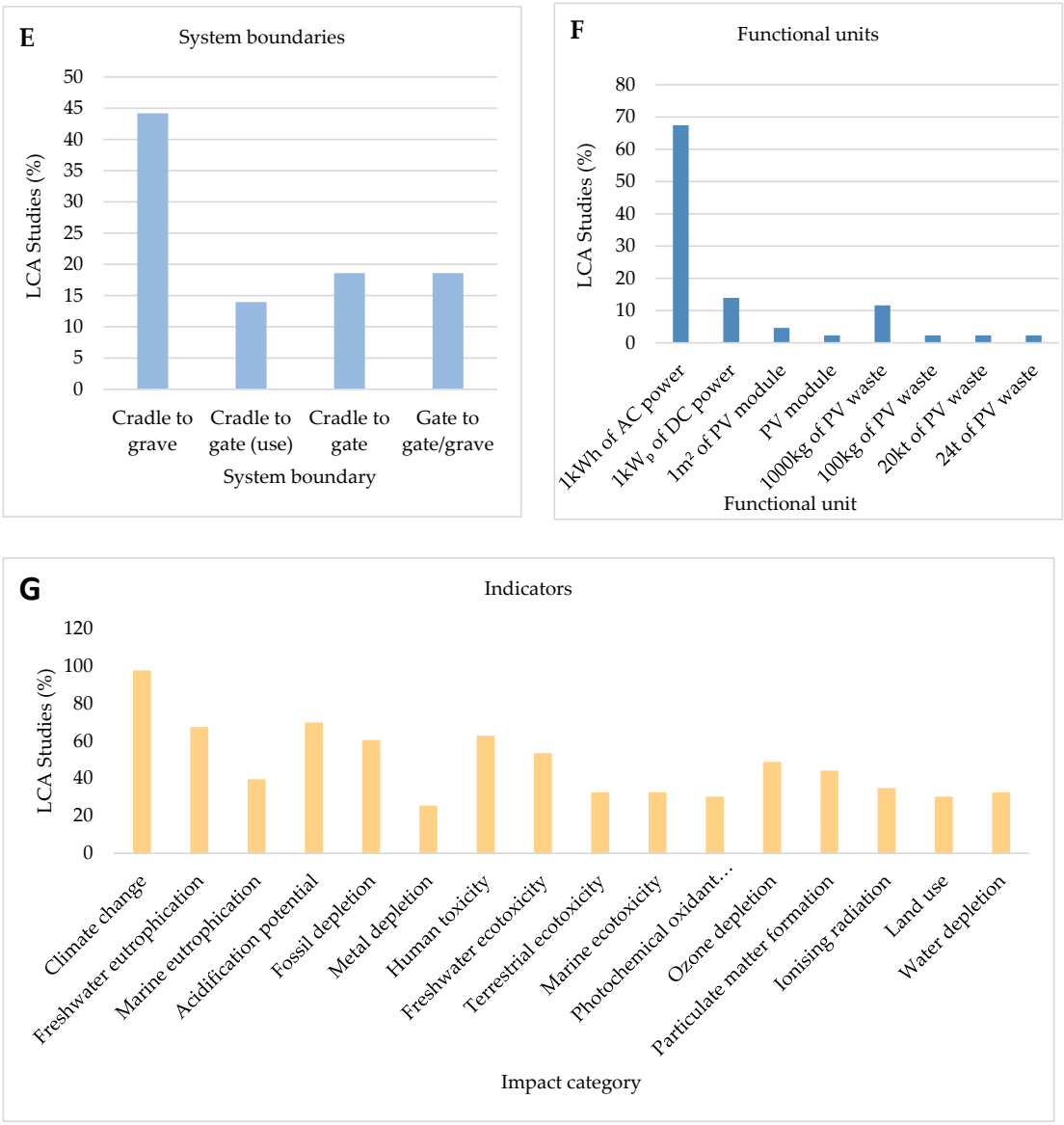


Figure 3. Graphical representation of previous LCA studies on solar modules (2015–2025).

3.2. Quality of Studies Considered for Data Extraction

Figure 4 presents the detailed evaluation of the selected studies across various methodological aspects. Concerning system boundaries, 7 studies were classified as high risk due to their limited scope (Gate-gate/grave) and 14 studies were considered moderate risk (cradle–gate), while the rest were regarded as low risk as they adopted the comprehensive cradle-to-grave approach. Regarding functional units, all the studies are deemed low-risk due to clearly defined functional units. Seventeen studies are identified as moderate risk due to reliance on secondary inventory data, while the rest are considered low risk due to the use of primary data. Studies with no clearly specified LCIA method were considered moderate risk (8), while the rest of the studies with clearly defined LCIA methods were considered low risk. All the studies are categorized as low risk with clearly reported impact categories. Regarding sensitivity analysis, studies without sensitivity analysis were considered moderate risk (16), while studies with such analysis were rated as low risk. Based on the above findings, 20 studies were classified as low risk, 16 as moderate risk, and 7 as high risk.

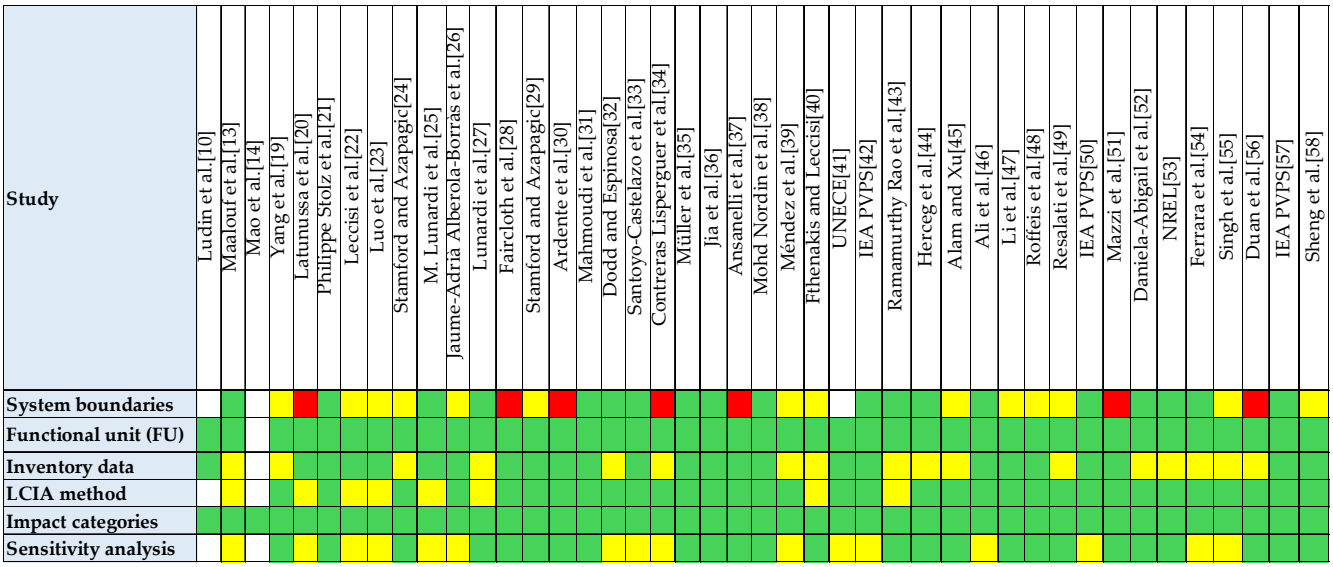


Figure 4. Quality of included studies (green: low risk; yellow: moderate risk; and red: high risk) [10,13,14,19–58].

3.3. Analysis of the Reviewed Studies

These sections below present the results of the review, including the statistical analysis of some environmental impact indicators of solar technologies, including climate change (GHG emissions), freshwater eutrophication, marine eutrophication, acidification potential, fossil depletion, metal depletion, human toxicity, freshwater ecotoxicity, terrestrial ecotoxicity, marine ecotoxicity, photochemical oxidant formation, ozone depletion, particulate matter formation, ionizing radiation, land use, and water depletion. The functional units (FUs) used in the reviewed studies are diverse, with kW_p of electricity, kWh of electricity, m^2 of cells, and kg of waste materials being the most used. However, for comparison between the different studies, FUs of m^2 have been converted to kW_p using the reference flows (m^2/kW_p) of the modules, while FUs of kWh have been converted to kW_p using Equation (1) below (GHG emissions example):

$$kgCO_2eq/kW = \frac{kgCO_2eq}{kWh} * \frac{(GTI \times Q \times L)}{I_{STC}} \quad (1)$$

where GTI is the yearly global irradiation on a tilted plane ($kWh/m^2/year$).

I_{STC} is the incident solar radiation under STC (1 kW/m^2).
Q is the performance ratio (5).
L is the system lifetime (years).

3.4. Climate Change/GHG Emissions

Climate change quantifies the impact of greenhouse gas (GHG) emissions from a product's entire life cycle. It is quantified as Global Warming Potential (GWP) expressed in kilograms of carbon dioxide equivalent ($kgCO_2$-eq). The GHG emissions of various types of solar modules and cells have been presented from the studies ranging from 2015 to 2025 in the sections below. The various studies on PV technologies categorized according to the country of publication/production are shown in Figure 5. Generally, c-Si PV technologies tend to exhibit higher GHG emissions compared to thin-film PV technologies. Leccisi et al. [22] indicate that among c-Si modules, those produced in China frequently report the highest GHG emissions. On the contrary, CdTe modules tend to show comparatively lower GHG emissions, particularly when production takes place in the U.S. Generally,

the GHG emissions are noticeably lower when manufacturing/production takes place in Europe in comparison with the USA and China. Predominantly, the Chinese production chain shows the highest GHG impact values due to the Chinese coal-dominated electricity mix, combined with the high-energy requirements for solar-grade silicon and c-Si module production. Production of c-Si modules in the United States and Europe, where electricity generation is less reliant on coal, can significantly reduce their GHG impacts, making them more comparable to thin-film modules [59]. A study by Brittany L. Smith et al. [53] also showed that modules imported from China and South-East Asia have higher GHG impacts than those manufactured in the U.S., with those in the U.S. low-carbon supply chain showing even lower emissions, credited to the cleaner U.S. electricity grid. It is important to note that the majority of c-Si PV modules are primarily manufactured in China, while thin films are mostly produced in Europe, the U.S., and Japan. It is important that results are cautiously interpreted, as differences in system boundaries, methodological choices, and supply chain systems significantly influence reported results.

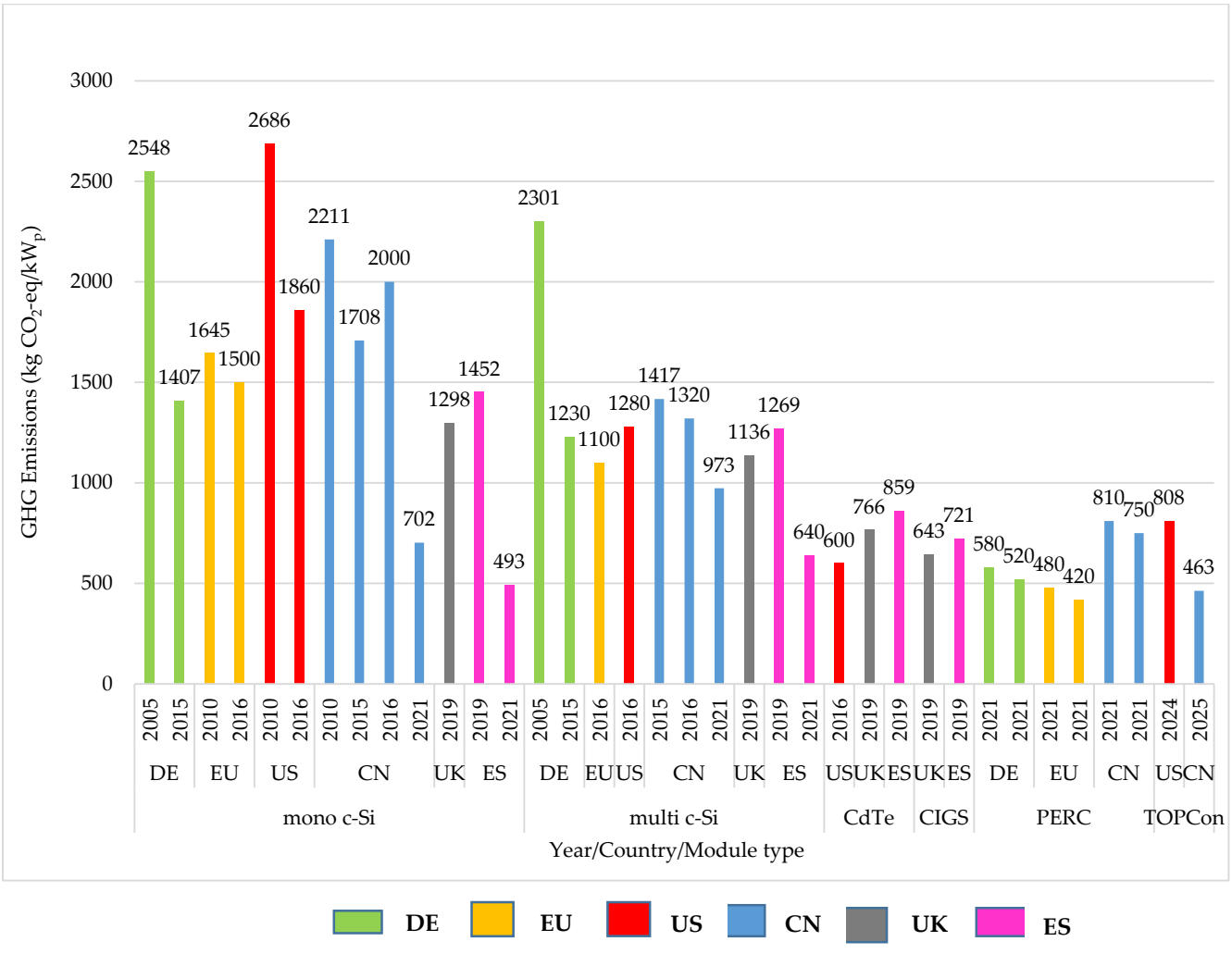


Figure 5. Analysis of PV system GHG emissions in regard to countries and years of study.

Geographical factors, such as solar irradiation, further influence life cycle emissions when results are expressed per unit of electricity generated (kWh). In their study, Stamford and Azapagic [29] revealed that the environmental impacts of c-Si and thin-film PV installations in the UK are 72% higher than in Spain, primarily due to the differences in solar irradiance between the two countries. For example, the GHG emissions for a CIGS installation are estimated at 25 g CO_2-eq/kWh in the UK compared to 14.6 g CO_2-eq/kWh

in Spain. However, when emissions are expressed per unit of installed capacity (kW_p) and normalized over comparable lifetimes, these differences become less pronounced (642.8 kg CO_2-eq/kW_p in the UK and 720.9 kg CO_2-eq/kW_p in Spain), highlighting the importance of the chosen functional unit in interpreting results. In another study, Müller et al. [35] compared GHG emissions of glass–backsheet (G-BS) and glass–glass (G-G) PERC modules produced in China, Germany, or the EU and found emissions of 810 and 750 kg CO_2-eq/kW_p for Chinese production, 580 and 520 kg CO_2-eq/kW_p for Germany, and 480 and 420 kg CO_2-eq/kW_p for EU production, respectively. This indicates that production in Germany and the EU leads to nearly 30% and 40% lower GHG emissions than in China. In addition, G-G modules demonstrate lower emissions compared to G-BS modules by 8%, 11%, and 12.5% in China, Germany, and the EU, respectively. In a related study, due to the higher carbon intensity of its electricity mix, GHG emissions from Chinese production of upgraded metallurgical-grade (UMG) silicon and polysilicon are 33% and 43% higher, respectively, than those in Spain [39]. When comparing pairs, the differences between UMG and polysilicon production in Spain and China are 24% and 15%, respectively, mainly due to the higher electricity and heat demand involved in polysilicon manufacturing. It is established that while clear trends are identified, such as the influence of energy mix, production location, and technological improvements, the variability across the studies highlights the need for careful interpretation due to differences in system boundaries, methodologies, and functional units.

This section analyzes the GHG emissions of various PV technologies for the study period of 2015–2025. For silicon technologies, GHG emissions from studies since 2015 for mono c-Si have been reported in the range of 493–2760 kg CO_2-eq/kW_p, as shown in Figure 6, which is relatively comparable to those for multi c-Si, which ranged between 640 and 2418 kg CO_2-eq/kW_p. However, these ranges should be interpreted with caution as differences in system boundaries and methodological approaches limit direct comparability across studies. Ludin et al. [10] attributed the relatively high GHG emissions for both silicon technologies in comparison to other solar PV module types to the highly energy-intensive processes associated with silicon purification and crystal growth. For thin-film technologies, CdTe generally exhibits lower GHG emissions, with a range of 167–2140 kg CO_2-eq/kW_p, primarily due to lower energy consumption during module production [10]. In contrast, a-Si (1876–2558 kg CO_2-eq/kW_p) and CIGS (262–2675 kg CO_2-eq/kW_p) display wider and generally higher emission ranges, reflecting differences in production processes. Regarding third-generation modules, PERC solar modules with emissions ranging from 425–1759 kg CO_2-eq/kW_p and the limited data available on TOPCon modules with emissions of 463 kg CO_2-eq/kW_p demonstrate an improving environmental performance commonly associated with reduced energy consumption and improved manufacturing efficiency. Ludin et al. [10] link the reduced environmental impacts of the new emerging technologies, such as DSSD and Perovskite cells, to their use of glass substrates and the reduction in solvent consumption and raw material waste, but the findings remain indicative as data for such technologies is still limited. Ludin et al. [10] discuss that the variation in GHG emissions across technologies can be attributed not only to technological differences but also to external factors, such as country-specific electricity mixes, solar irradiation levels, and lifetimes of PV technologies. The results in Figure 5 clearly show an environmental improvement for mono c-Si and multi c-Si PV systems, with their upper bound values decreasing by approximately 35% and 37%, respectively, compared to earlier results from 2020–2022. Fthenakis and Leccisi [40] attributed the substantial reductions to the enhanced material efficiency during solar-grade silicon and wafer manufacturing stages.

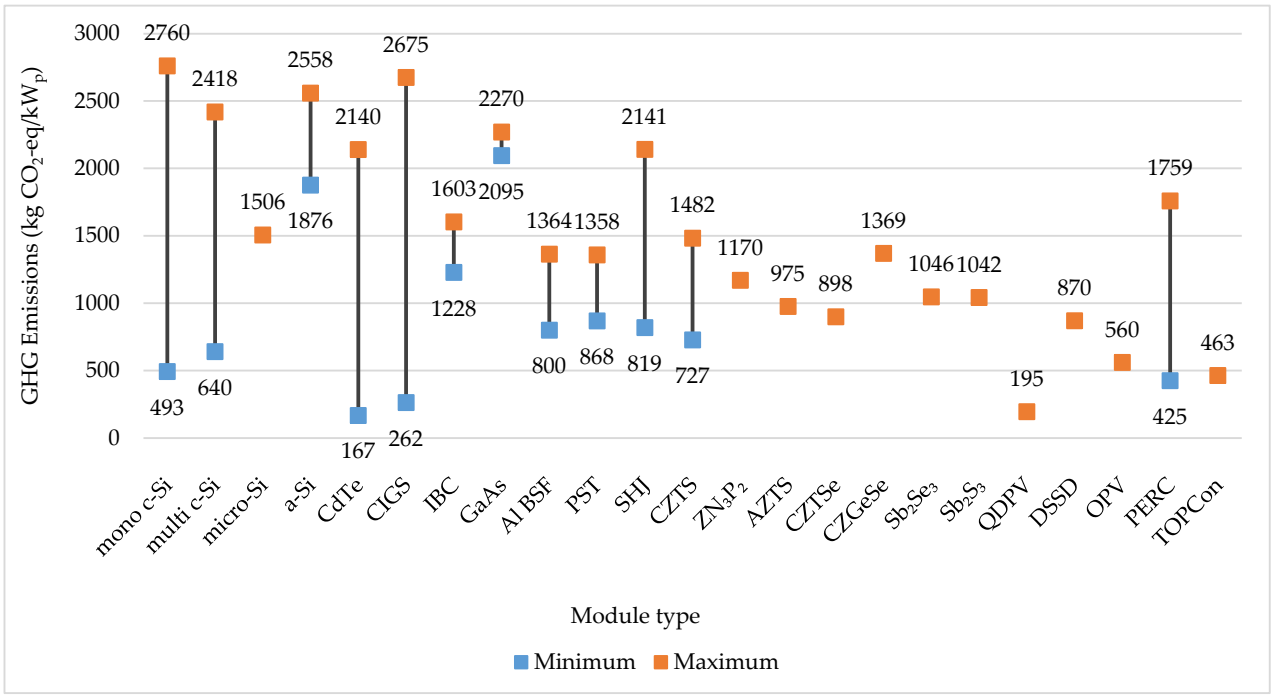


Figure 6. Analysis of PV system GHG emissions in regard to previous studies (2015–2025).

A study by Stamford and Azapagic [24] reported moderate reductions in the environmental impact of PV systems ranging between 8% and 34% during the 2015–2025 period. These reductions were lower than those observed between 2005 and 2015, when more substantial improvements were recorded. The reduced rate of improvement between 2015 and 2025 was largely attributed to the dominance of Chinese-manufactured PV systems, which generally have higher environmental burdens, thus lessening the improvement rate. However, this study did not account for the rise of emerging disruptive PV technologies or potential improvements in the Chinese energy mix during that period. In contrast, Herceg et al. [44] highlighted a consistent decline in environmental impacts from 2004 to 2020 due to technological advances in PV manufacturing, like the use of thinner wafers and improved module efficiencies. Despite these general trends, some studies have reported substantially higher GHG emissions. For instance, Jaume-Adrià Alberola-Borràs et al. [26], who conducted a cradle-to-gate LCA of a perovskite module based on the carbon mesoporous stack architecture and produced via a pre-industrial process, reported GHG emissions of 4207 kg CO_2-eq/kW_p, with approximately 50% attributed to the energy-intensive production of the glass substrate. Similarly, Santoyo-Castelazo et al. [33] reported the GHG emissions of 3541 kg CO_2-eq/kW_p for mono c-Si modules mainly due to the release of CO_2, N_2O, and traces of CH_4 linked to fossil fuel-based electricity use. It is important to note that such high estimates often result from broader system boundaries, such as cradle-to-grave assessments, which include BOS components and additional life cycle stages, thereby capturing a more comprehensive but not directly comparable environmental profile.

This section discusses the environmental impacts and benefits of recycling PV modules based on previous studies. Daniela-Abigail et al. [52] credit the reduction in environmental impacts through recycling to the reduced demand for raw material extraction and processing, but in order to maximize these benefits, there is a need for strategic planning to determine optimal distances for transporting the panels to minimize emissions during their disposal, recycling, and/or installation. In their paper, Latunussa et al. [20] present the FRELP process as one of the most advanced recycling methods studied. Recycling 1000 kg

of PV waste panels through this process produced an overall climate change (CC) impact of 370 kg CO_2-eq, mainly due to PV waste transportation, incineration, and metal recovery operations such as sieving, acid leaching, electrolysis, neutralization, and filtration. At the same time, environmental credits of −75.90 kg CO_2-eq were achieved through energy recovery from the generation of heat and electricity equivalent to that from alternative systems. Subsequent studies [14,28,30,51] also applied the same process and reported net negative GHG emissions of −1939, −4500, −1481, and −660 kg CO_2-eq, respectively. In a comparison between the FRELP process and other methods, like LGRF and landfilling, Faircloth et al. [28] indicate that although the FRELP process requires greater electricity and material inputs, its high recovery rates for aluminum, copper, and glass, in addition to silicon and silver, are able to displace enough primary production of those materials and offset the environmental burdens.

Among the recovered secondary materials, aluminum provides the greatest environmental benefit, followed by silver and silicon, reflecting their high embodied energy in primary production. The significant role of transport-related impacts observed by Latunussa et al. [20] was also highlighted in a study by Ansanelli et al. [37] on the ReSiELP process of recycling (treatment of 24 tons) EoL c-Si PV panels, demonstrating that transport logistics can significantly influence overall results, sometimes offsetting gains from material recovery. However, in many cases, the environmental benefits associated with avoided virgin material production exceed the burdens of recycling operations. In their study, Ansanelli et al. report that the environmental benefits due to recovery of the materials clearly exceeded the life cycle burdens of the recycling operations, as observed in the climate change category with a net impact of −136 kg CO_2-eq. Similar conclusions were reported by Mahmoudi et al. [31], whereby the GHG emissions from full recovery of EoL PV panels were considerably lower than those associated with primary production of c-Si PV modules using virgin inputs, achieving a net climate change benefit of −18,550 kg CO_2-eq in one of the full-recovery scenarios.

In a similar comparison between recycling and virgin materials extraction, it is evident that recycling becomes environmentally unjustifiable when the environmental impacts from recovery processes surpass those from extracting raw materials. Maani et al. [60] compared the environmental impacts of several recycling processes for c-Si and CdTe modules against virgin material extraction (glass, Al, and Si for c-Si and glass for CdTe). The results showed that all evaluated recycling methods for c-Si modules resulted in higher environmental burdens than virgin material extraction, suggesting that the current recycling methods remain environmentally unsustainable. The use of these secondary raw materials in the remanufacturing of PV modules has been shown to significantly reduce GHG emissions, particularly when secondary silicon replaces primary silicon. A study by Salvi et al. [61] shows that recovering only metallurgical grade silicon leads to only 17% reduction in GHG emissions, as substantial energy is still required to refine MG-Si into SG-Si. Given that solar cell production represents the largest source of emissions in the PV supply chain, the use of secondary silicon in this phase becomes particularly impactful. Glass production ranks as the second largest emission source, followed by transportation.

The integration of renewable energy throughout the manufacturing process and the implementation of circular economy approaches, such as recycling, can further reduce environmental impacts, as the emissions resulting from recycling processes are relatively insignificant compared to the savings achieved through lower energy consumption when using recycled materials. Similarly, Contreras Lisperguer et al. [34] establish that closed-loop recycling and material recovery can significantly reduce environmental impacts, achieving up to 74% GHG emission reductions despite some residual impacts from trans-

portation, plastic incineration, and landfilling of sludge and fly ash. Several of the benefits of recycling reported in the literature are illustrated in Figure 7.

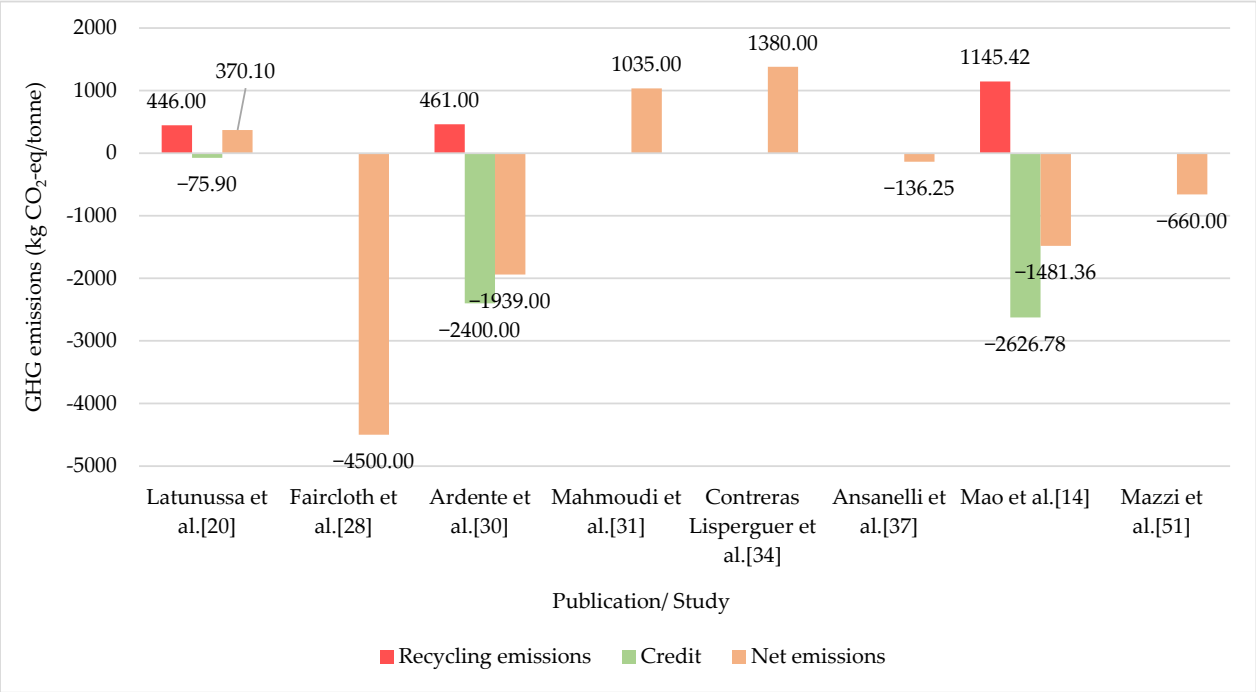


Figure 7. Analysis of PV system GHG recycling emissions [14,20,28,30,31,34,37,51].

3.5. Eutrophication Potential (EP)

Eutrophication potential describes the potential environmental damage resulting from the release of excess nutrients like nitrogen (N) and phosphorus (P) into ecosystems. It can be further classified into freshwater eutrophication, in which phosphorus is the limiting nutrient, and marine eutrophication, in which nitrogen is the limiting nutrient. Freshwater eutrophication potential (FEP) is quantified in phosphorus equivalents (P-eq) while marine eutrophication potential (MEP) is quantified in nitrogen equivalents (N-eq). The freshwater eutrophication impacts of various solar module and cell technologies reported between 2015 and 2025 are summarized here. For silicon technologies, the eutrophication emissions from previous studies for mono c-Si are reported in the range of 0.19–6.22 kg P-eq/kW_p, as shown in Figure 7, which are comparable to those of multi c-Si, ranging from 0.16–5.04 kg P-eq/kW_p. Similar to climate change impacts, the higher eutrophication emissions in silicon technologies are mostly linked to the high-energy consumption associated with silicon purification and crystal growth, which indirectly contribute to nutrient emissions (phosphorus and nitrogen) that occur during the upstream power generation and fuel supply processes, including emissions associated with fuel extraction, combustion, and ash handling, as implied by Johnson et al. [62], who alludes to the big role played by phosphorus in accelerating the eutrophication of freshwater systems. For thin-film technologies, CdTe modules exhibit comparatively low eutrophication emissions, ranging from 0.09–1.4 kg P-eq/kW_p, primarily due to lower energy requirements during module production. Similarly, a-Si (0.09–0.13 kg P-eq/kW_p) and CIGS (0.06–1.37 kg P-eq/kW_p) show relatively low emission ranges. For emerging solar cells, such as PERC cells, emissions fall between 0.22 and 0.90 kg P-eq/kW_p, demonstrating an improving environmental performance linked to reduced energy demand.

Results from more recent studies (2024) clearly show an environmental improvement for mono c-Si and multi c-Si PV systems, with maximum eutrophication values decreasing

by up to 90% and 89%, respectively, between 2020 and 2024 as shown in Figure 8. Fthenakis and Leccisi [40] attribute these substantial emission reductions to the enhanced material efficiency during solar-grade silicon and wafer fabrication stages. Philippe Stolz et al. [21] credit the low emission values to the potential recycling benefits owing to the avoidance of primary copper production, which reduces emissions from copper mining. However, considerable variability remains across studies due to the diversity of methodologies and system boundaries. For example, a study by Jaume-Adrià Alberola-Borràs et al. [26] that analyzed the environmental performance of a perovskite module from cradle to gate reports emissions as high as 11.24 kg P-eq/kW_p. Similarly, Santoyo-Castelazo et al. [33] report emissions between 9.58 and 11.58 kg P-eq/kW_p across multiple PV technologies, mainly due to inorganic emissions released into freshwater ecosystems by pollutants such as phosphates, total organic carbon, and nitrogen oxides during the manufacturing process. In contrast, M. Lunardi et al. [25,27] reported significantly lower eutrophication values, ranging between 7.80 × 10^{-7} and 1.37 × 10^{-6} for Al-BSF, PERC, and Tandem LSC-Si module technologies. It is important to note, however, that BOS components and transportation were excluded in these LCAs, likely resulting in an underestimation of the total environmental burdens.

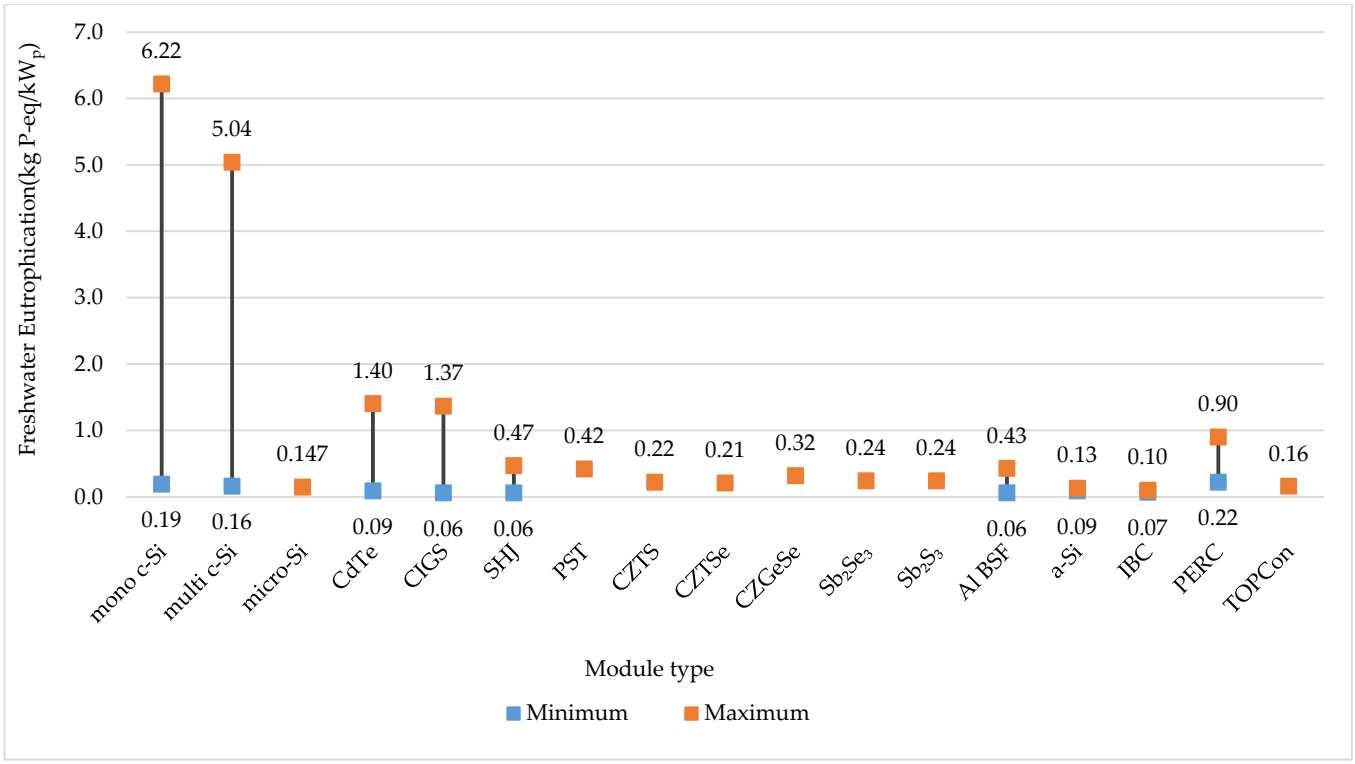


Figure 8. Analysis of PV system FEP in regard to previous studies (2015–2025).

Concerning the eutrophication impacts of PV recycling, Latunussa et al. [20] found that the treatments for metal recovery stages, which include sieving, acid leaching, electrolysis, and acid neutralization from the bottom ash, are the most influential for freshwater, marine, and terrestrial eutrophication impacts. Conversely, the use of recycled materials can provide notable environmental benefits; for example, in the FRELP process by Ardente et al. [30], recycled silver, silicon, and aluminum yield the greatest environmental benefits for eutrophication potential, with recycled silver showing significant benefits in reducing freshwater eutrophication impacts. However, the environmental burdens have been found to arise from the use of energy-intensive thermal treatments and fossil fuel-based transportation during the recycling process [35,52]. According to Contreras Lisperguer et al. [34], these

environmental impacts are linked to pyrolysis, fossil fuel-based transportation, incineration of plastics, landfilling of PV residues, and energy-intensive fossil-based thermal treatments used at the end of life. The FRELP process applied in [14,28,30,37,51] assessed the environmental impacts associated with the recycling of 1000 kg c-Si PV waste, leading to net negative freshwater eutrophication impacts of −1.11, −1.546, −2.3, −1.11, and −0.32kg P eq, respectively as presented in Figure 9, indicating that PV recycling provides a net environmental benefit. However, the magnitude of these benefits varies based on the efficiency of material recovery, energy sources used, and transportation, underscoring the importance of carefully designed recycling systems.

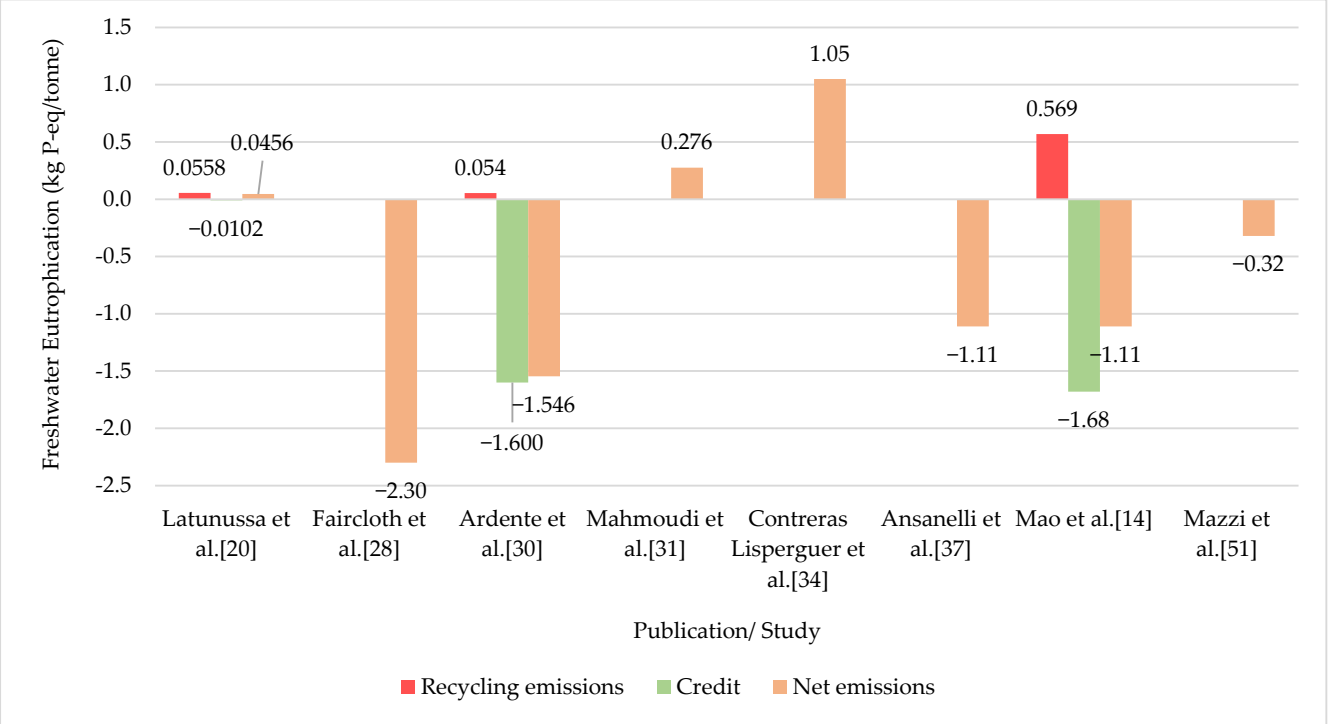


Figure 9. Analysis of PV system FEP recycling emissions [14,20,28,30,31,34,37,51].

3.6. Acidification Potential (AP)

Acidification potential measures the ability of the product or process emissions to contribute to acid rain, thus acidifying soils and waterbodies and bringing harm to the ecosystems. Depending on the chosen impact assessment methods, acidification is quantified in sulfur dioxide equivalents (SO_2-eq) or moles H+ equivalent (mol H+-eq), with SO_2-eq being the most commonly used unit. It is applied in CML and ReCiPe methods to normalize the impacts of various acidifying pollutants, like nitrogen oxides and sulfur dioxides, into a single comparable metric. The acidification potential of various solar modules and cell types from studies conducted between 2015 and 2025 is summarized here. For silicon technologies, the acidification emissions from previous studies for mono c-Si are reported in the range of 3.25–29.10 kg SO_2-eq/kW_p, as shown in Figure 10, which are quite similar to those of multi c-Si, which range from 4.09–30.00 kg SO_2-eq/kW_p. Analogous to climate change and eutrophication, the energy-intensive silicon refinement and crystal growth stages primarily contribute to the higher acidification emissions for silicon technologies due to the air pollutants that are emitted during the upstream power generation and fuel supply processes, including emissions of sulfur dioxide and nitrogen oxides [63]. For thin-film technologies, CdTe generally exhibits lower acidification emissions with a range of 2.7–12.28 kg SO_2-eq/kW_p due to the relatively low energy requirements dur-

ing the module's production, in contrast to a-Si (11.94–27.72 kg SO_2-eq/kW_p) and CIGS (3.44–22.85 kg SO_2-eq/kW_p), which have wider and more variable emission ranges. Emerging solar cells, such as PERC cells, demonstrate comparatively lower emissions ranging from 2.51–10.23 kg SO_2-eq/kW_p, indicating an improved environmental profile, largely due to lower energy consumption during cell production. More broadly, according to Ludin et al. [10], newer emerging technologies may achieve lower environmental impacts due to innovations such as the use of the glass substrate and minimized solvent and raw material waste, although this is dependent on specific production scenarios.

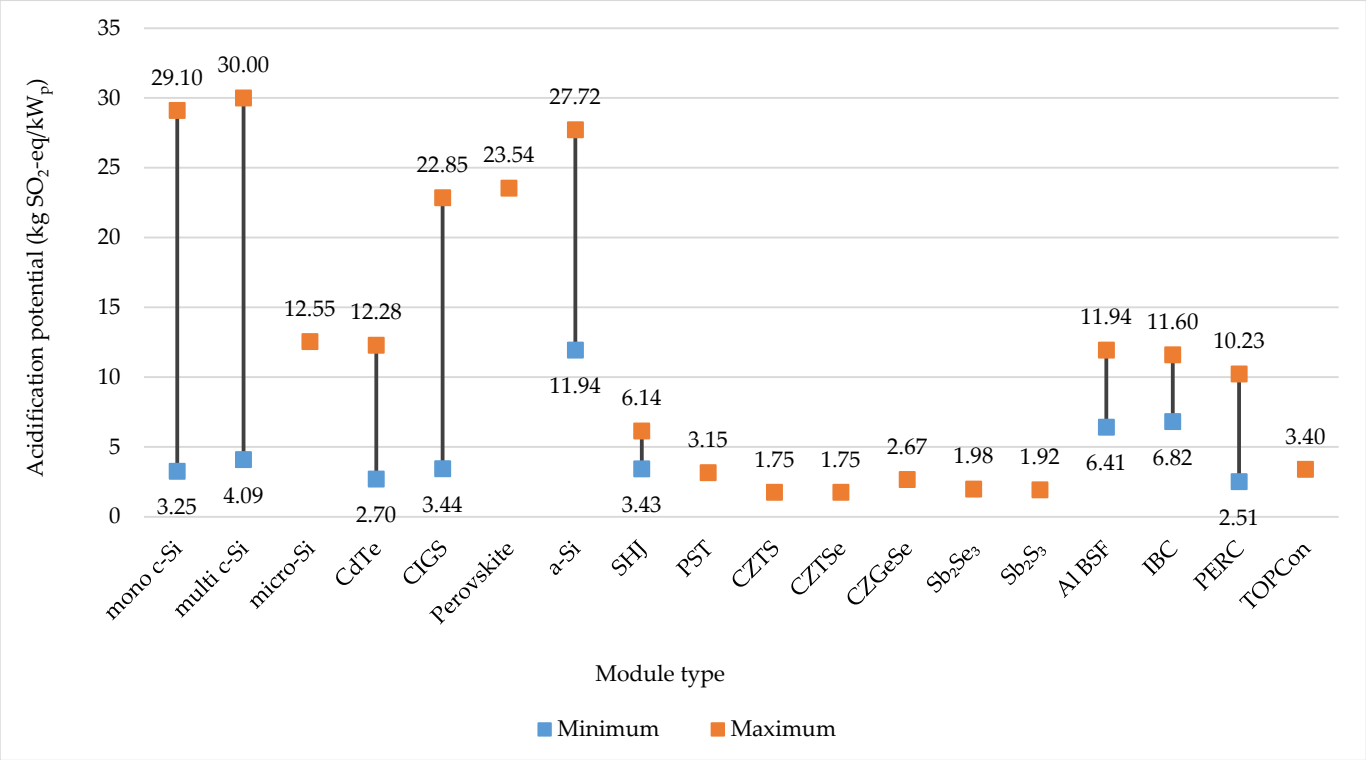


Figure 10. Analysis of PV system AP in regard to previous studies (2015–2025).

The results of previous studies (2024) indicate a general trend of decreasing acidification impacts for mono c-Si and multi c-Si PV systems, with the maximum acidification values reduced by as much as 61% and 48%, respectively, between 2021 and 2024. Fthenakis and Leccisi [40] attribute the substantial emission reductions to improved material efficiency during solar-grade silicon and wafer manufacturing stages. However, substantial variability persists across studies. Méndez et al. [39] attributed the high emission values for multi c-Si to variations caused solely by the differences in the energy mix used for calculations, with results leading to values over 4.5 times higher for the Chinese scenario compared to that of Spain. Furthermore, Santoyo-Castelazo et al. [33] attributed the high emission values for mono c-Si modules to the manufacturing stage in which inorganic emissions into the air, such as sulfur dioxide, nitrogen oxide, and some ammonia, were emitted during multi-Si wafer and PV panel manufacturing.

Concerning the acidification impacts of PV recycling, Latunussa et al. [20] indicate that metal recovery treatments, including sieving, acid leaching, electrolysis, and acid neutralization from the bottom ash, and NOx emissions during electrolysis are the most impactful for acidification emissions. Similarly, Contreras Lisperguer et al. [34] indicate that the environmental burdens stem from pyrolysis, fossil fuel-based transportation, plastic incineration, landfill of PV residues, and energy-intensive thermal processes reliant on

fossil fuels. Despite these burdens, the FRELP process applied in [14,28,30,37,51] for the treatment of 1000 kg c-Si PV waste reported net negative acidification emissions of −2.89, −12.4, −36, −28.81, and −2.7 kg SO_2 eq, respectively as shown in Figure 11, indicating that PV recycling is environmentally beneficial in terms of acidification potential due to the avoidance of primary material production.

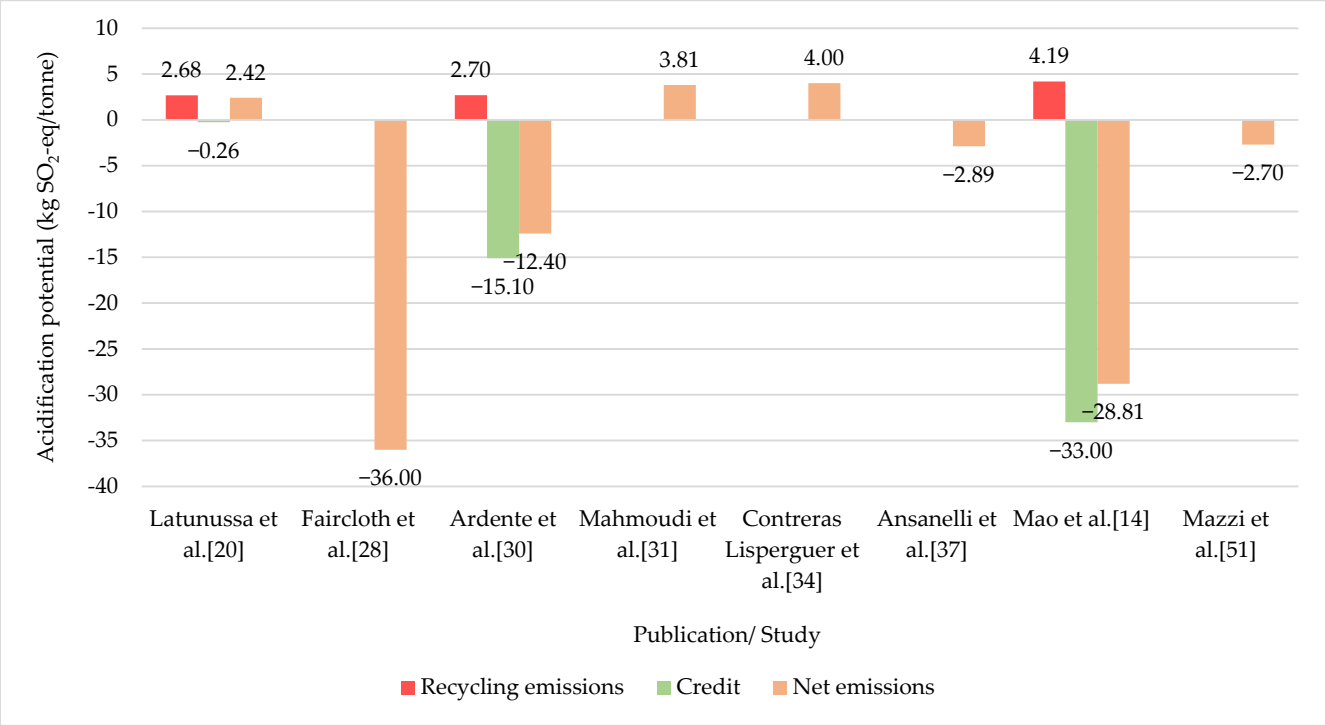


Figure 11. Analysis of PV system AP recycling emissions [14,20,28,30,31,34,37,51].

3.7. Fossil Depletion Potential (FDP) and Metal Depletion Potential (MDP)

Fossil depletion refers to the exhaustion of non-renewable fossil fuels like coal, oil, and natural gas, which are consumed faster than they regenerate. The terms fossil depletion and fossil resource depletion are used interchangeably; however, they are synonymous terms. Depending on the impact assessment method, fossil depletion is either quantified in kg oil eq, as used in ReCiPe methods, or as abiotic depletion potential measured in kg Sb-eq, as commonly used in CML methods. Most studies express fossil depletion in kg Sb-eq, and the results are summarized here. Previous studies report relatively low emission values ranging from 0.007 to 0.564 kg Sb-eq/kW_p across different solar PV technologies as indicated in Figure 12. The observed trends in climate change, eutrophication, and acidification described above can be explained by FDP since it highlights electricity demand as a dominant factor in environmental performance; therefore, the energy-intensive silicon refinement and crystal growth stages result in higher fossil resource use for silicon technologies in comparison to other solar PV module types [64]. Despite the generally low ranges reported, some studies indicate substantially higher fossil depletion impacts. One study by Philippe Stolz et al. [21] reported notably high emissions for the CIGS PV system at 14.98 kg Sb-eq/kW_p. Similarly, Rabaia et al. [65] presented relatively high values compared to other studies for multi-c-Si modules, reaching 3.4 kg Sb-eq/kW_p. The variability across studies reflects the importance of consistent methodological choices and careful interpretation of results.

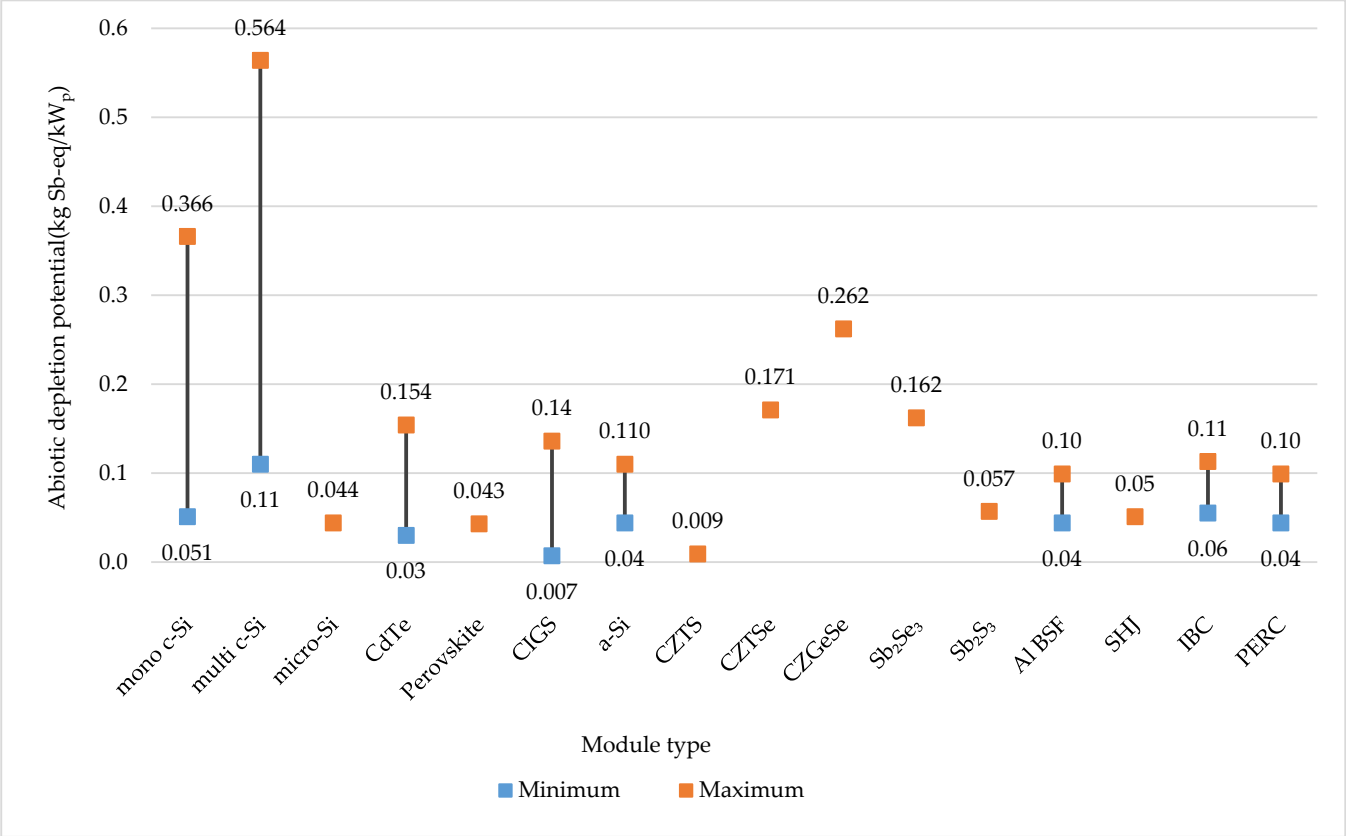


Figure 12. Analysis of PV system FDP in regard to previous studies (2015–2025).

Regarding the fossil depletion impacts of PV recycling, studies indicate that the main environmental burdens are primarily driven by transportation and electricity consumption associated with recycling operations [37]. According to Mazzi et al. [51], most of the impacts in this category arise from thermal treatment processes. Similarly, Contreras Lisperguer et al. [34] mention that the environmental burdens in this category are mainly due to the use of fossil fuel-based thermal operations and transportation during the recycling process. In spite of these burdens, the FRELP process applied in [14,28,30], which assessed the environmental impacts associated with the disposal of 1000 kg c-Si PV waste, reports net negative emission values of −4.796 in kg Sb eq, −950, and −504.31 in kg oil eq, respectively as shown in Figure 13, indicating the benefits of PV recycling.

Metal depletion assesses the environmental impact of the extraction and use of mineral and metal resources. Similar to fossil depletion, it can also be assessed using abiotic depletion potential measured in kg Sb-eq. It is also quantified in kg Cu-eq or kg Fe-eq. depending on the impact assessment method. A limited number of studies have reported metal depletion and are mostly quantified in kg Cu-eq, as presented here. The results from these studies shown in Figure 14 fall within the same range of 1.24 to 16.51 kg Cu-eq for the different solar PV technologies. Generally, silicon technologies tend to exhibit lower metal depletion impacts due to the relative abundance of silicon. However, most impacts arise from the use of silver, aluminum, and copper. In contrast, Helbig et al. [66] indicate that thin-film technologies show higher metal depletion impacts because they depend on scarce by-product metals such as tellurium, indium, and selenium. Overall, while metal depletion impacts are less frequently reported, the available evidence suggests that the use of scarce metals plays a big role in determining the resource-related environmental performance of PV technologies.

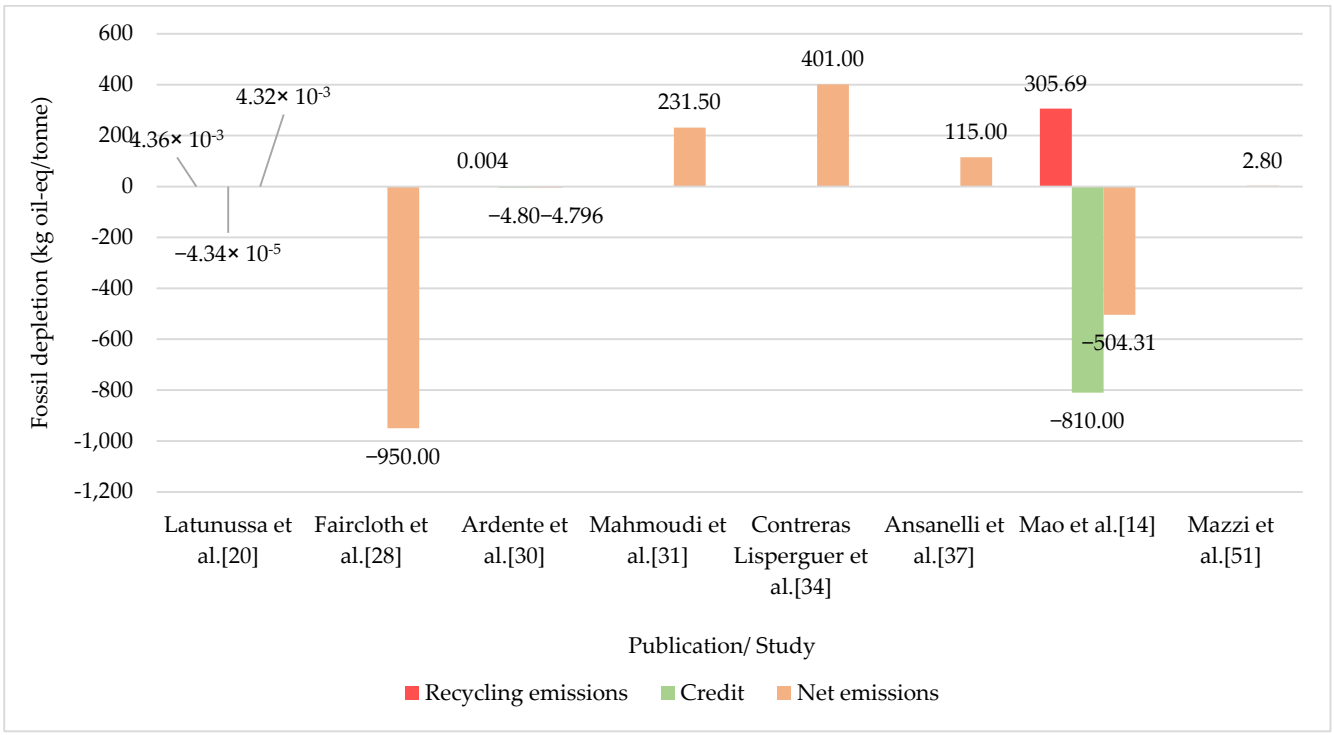


Figure 13. Analysis of PV system FDP recycling emissions [14,20,28,30,31,34,37,51].

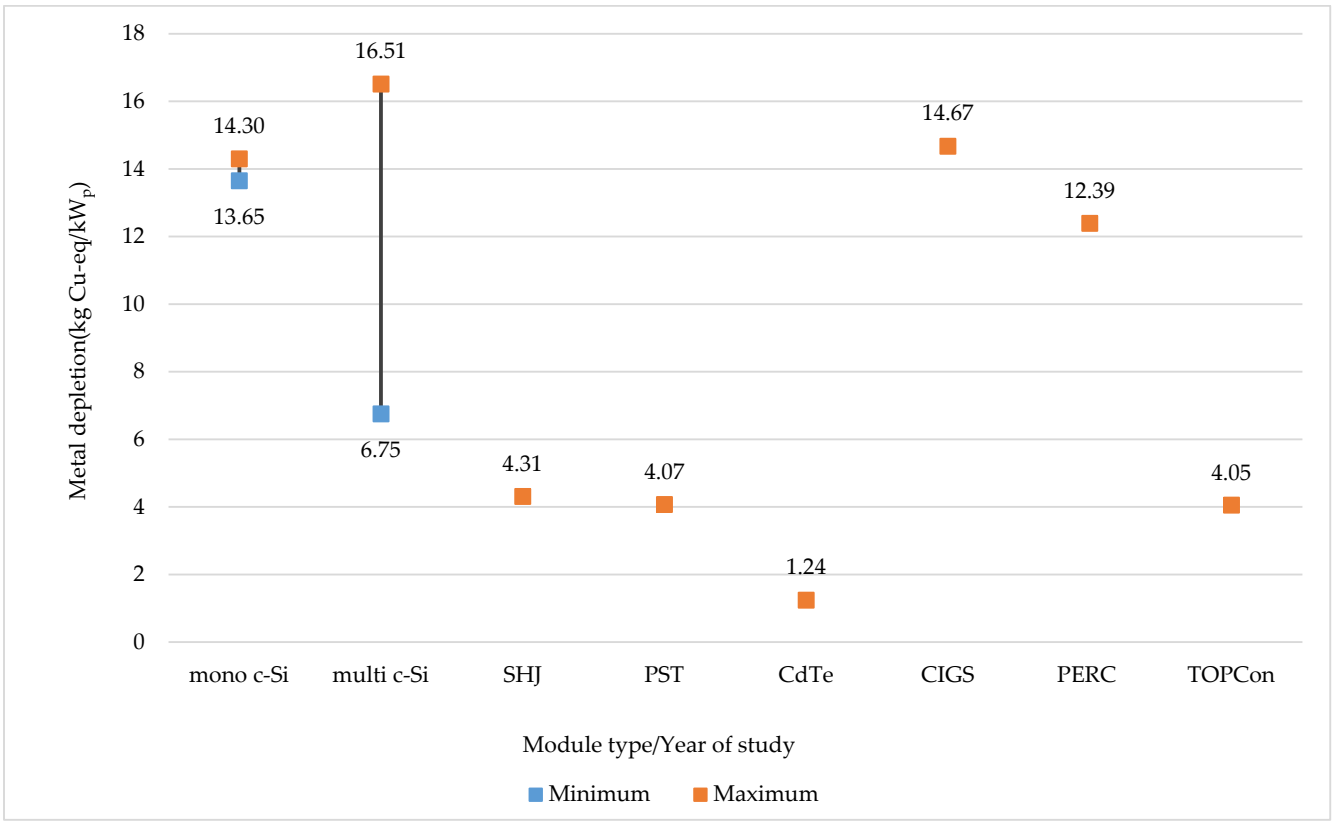


Figure 14. Analysis of PV system MDP in regard to previous studies (2015–2025).

For the metal depletion impacts from PV recycling, it is not possible to fully recover all the metals during the recycling process, increasing the environmental burden in this category. Contreras Lisperguer et al. [34] indicate that the low recovery rate of aluminum

from bottom ash, combined with the loss of other metals such as copper, silicon, tin, and lead due to thermal degradation, further contributes to higher emissions in this impact category. This study also notes that the environmental burdens in this category arise from fossil fuel-based thermal processes and transportation. However, the FRELP process applied in [28,37,51], which evaluated the environmental impacts associated with the treatment of 1000 kg c-Si PV waste, led to net emissions of −870 kg Fe eq, −64.2, and −3 kg Cu eq, respectively as shown in Figure 15, showing the environmental benefits of PV recycling in reducing metal resource depletion.

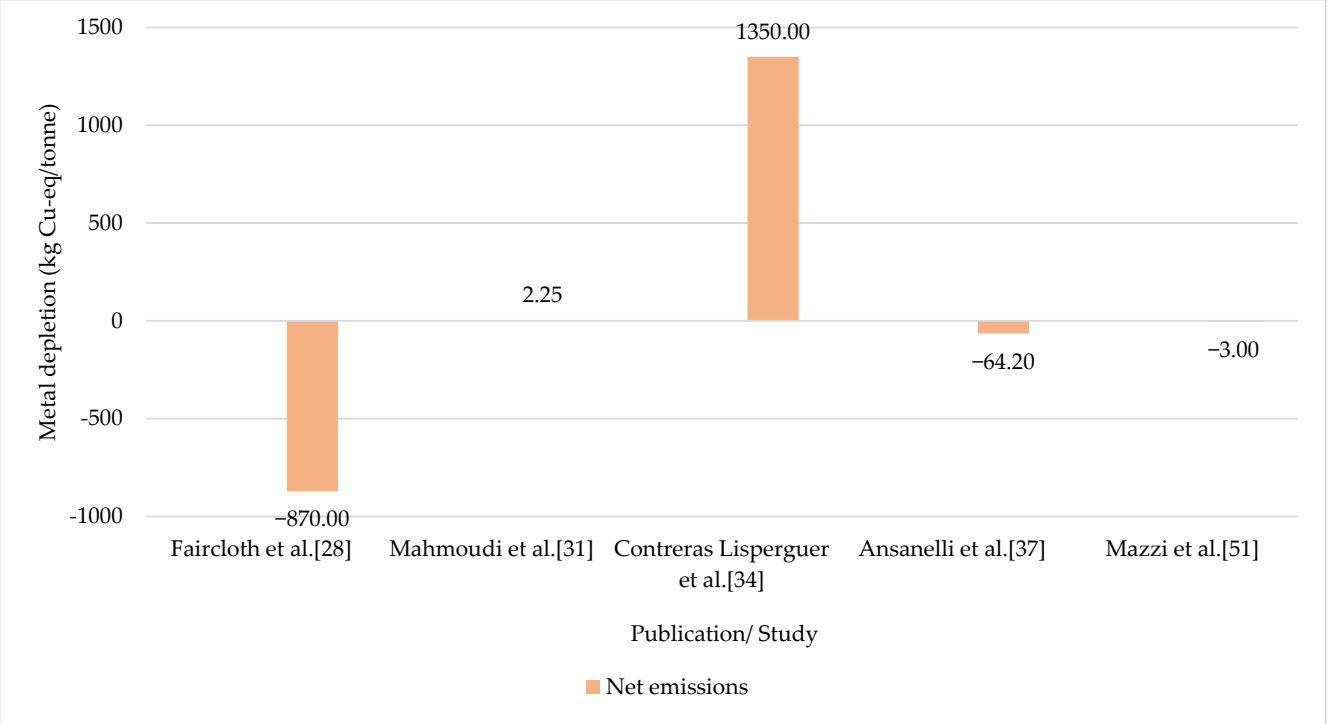


Figure 15. Analysis of PV system MDP recycling emissions [28,31,34,37,51].

3.8. Toxicity

Toxicity includes the impacts on both ecosystems (ecotoxicity) and human health (human toxicity) resulting from the persistence and accumulation of harmful substances in the environment, causing the increasing exposure of species and humans to toxic substances and eventually causing disease and death. Depending on environmental impact assessment methods, toxicity is measured in CTU (CTUe for ecotoxicity and CTUh for human toxicity), kg 1,4-DCB-eq, PAF/($m^2 \times$day), PAF/($m^3 \times$year), cases, or DALY. Among these units, CTU and kg 1,4-DCB-eq are the most widely used. The unit of kg 1,4-DCB eq is applied in CML and ReCiPe methods to normalize the impacts of different chemicals relative to 1,4-dichlorobenzenes. The following section describes the human toxicity (HT) of solar modules with units in kg 1,4-DCB-eq. For silicon technologies, the range of toxicity emissions from previous studies between 2015 and 2025 as highlighted in Figure 16 for mono c-Si has been reported as between 18.4 and 135 kg 1,4-DCB-eq/kW_p, which is broadly comparable to multi c-Si, which ranged from 32–138 kg 1,4-DCB-eq/kW_p, although the values should be interpreted with caution due to differences in system boundaries and methodological choices across studies. For thin-film technologies, CdTe exhibits lower human toxicity impacts, with a range of 27−108 kg 1,4-DCB-eq/kW_p. In contrast, CIS (79–187 kg 1,4-DCB-eq/kW_p) exhibits generally higher and more variable impacts. Regarding third-generation modules, PERC solar modules with emissions of 71 kg 1,4-DCB-

eq/kW_p demonstrate an improved environmental performance. It has been shown to be even lower for the TOPCon module, with emissions of 0.65 kg 1,4-DCB-eq/kW_p, although such values should be interpreted carefully. Joël Tchognia Nkuissi et al. [67] imply that human toxicity impacts are mainly driven by emissions of hazardous substances associated with material extraction and processing, which is generally high for thin-film technologies due to the use of toxic and scarce metals, such as cadmium, in their supply chains, while it is generally moderate for silicon technologies, mostly arising from upstream electricity generation and chemical use during silicon refinement.

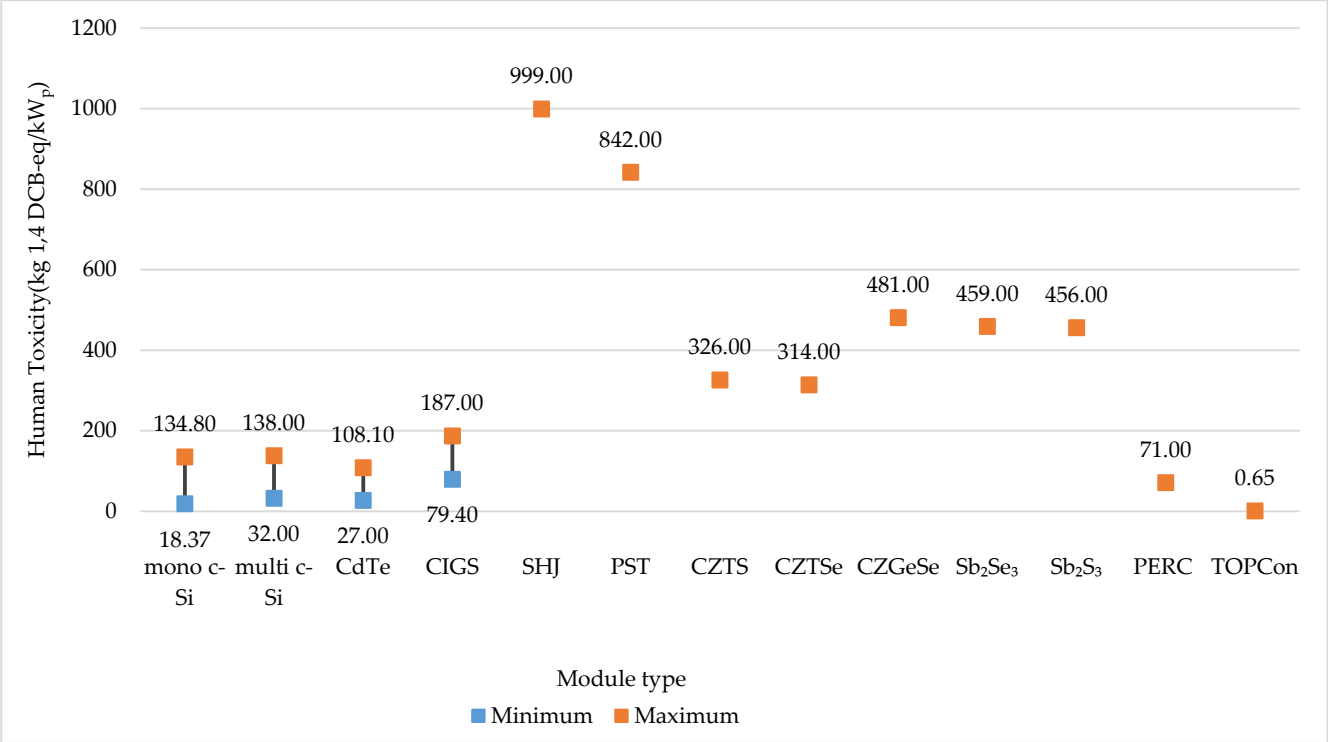


Figure 16. Analysis of PV system HT in regard to previous studies (2015–2025).

Given the diversity and variability of the studies, some studies have reported substantially higher figures. A study by Santoyo-Castelazo et al. [33] reported toxicity emissions ranging from 6764 to 7993 kg 1,4-DCB-eq/kW_p across various PV technologies, largely attributed to the emissions of certain heavy metals into the air, like arsenic, nickel, chromium, and cadmium, during the manufacturing process. Similarly, Stamford and Azapagic [24] report emissions between 3101 and 2777 kg 1,4-DCB-eq/kW_p for mono c-Si and multi c-Si PV systems, primarily resulting from emissions from metal production chains. Their analysis covered the entire life cycle from cradle to grave, capturing a more comprehensive but not directly comparable environmental profile.

Considering the toxicity impacts from PV recycling, studies indicate that certain stages of the recycling process contribute significantly to toxicity. Latunussa et al. [20] consider the incineration of PV sandwich layers and cable plastics, as well as the disposal of fly ash in hazardous waste landfills, to be the most impactful for freshwater ecotoxicity and human toxicity. However, for terrestrial ecotoxicity, the literature suggests that impacts are mostly caused by transportation and the use of chemical agents. It is mentioned that the environmental impacts associated with toxicity are mainly due to the toxic emissions generated during the recycling process, the use of hazardous chemicals for silicon recovery, and the incineration of the EVA and plastics, significantly increasing toxic impacts [31,34]. Notwithstanding these burdens, the FRELP process applied in [14,28,37,51] demonstrated

net negative emissions of −380, −3400, −2963, and −630 kg 1,4 DCB eq, respectively as shown in Figure 17, for the assessment of the environmental impacts associated with the disposal of 1000 kg c-Si PV waste, making PV recycling environmentally beneficial.

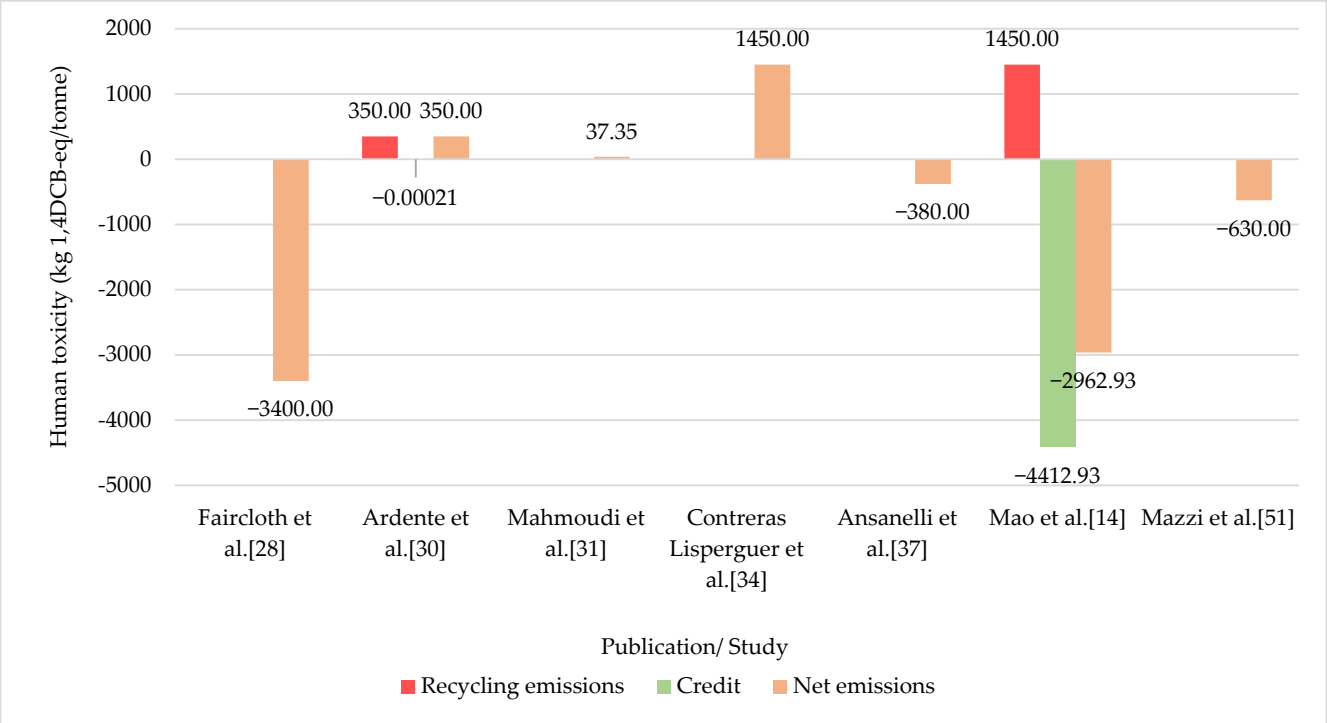


Figure 17. Analysis of PV system HT recycling emissions [14,28,30,31,34,37,51].

3.9. Photochemical Oxidant Formation Potential (POFP)

Photochemical oxidant formation quantifies the potential for a product or process to generate harmful atmospheric oxidants, primarily ground-level ozone, by quantifying the relative ozone-forming potential of Volatile Organic Compounds (VOCs) and Nitrogen Oxide (NOx) emissions. Its impacts on human health are quantified in either kilograms of ethylene equivalent (kg C_2H_4-eq) or kilograms of non-methane volatile organic compounds equivalent (kg NMVOC-eq). The following section describes the impact of photochemical oxidants expressed in kg NMVOC-eq for the different module technologies. The studies indicate generally consistent results, ranging between 0.10 and 9.55 kg NMVOC-eq/kW_p in Figure 18. Babaee et al. [63] consider the emissions of nitrogen oxides and volatile organic compounds that occur during the upstream power generation and fuel supply processes to be the reason why silicon modules exhibit higher POFP impacts compared to thin-film modules.

Some studies suggest higher values under specific conditions; for example, a study by Jaume-Adrià Alberola-Borràs et al. [26] on the environmental performance of a perovskite module, based on the carbon mesoporous stack architecture and produced with a pre-industrial process and analyzed from cradle to gate, reported higher emissions of 19.3 kg NMVOC-eq/kW_p, exceeding those reflected in other studies.

Concerning the photochemical oxidant formation impacts of PV recycling, Latunussa et al. [20] indicate that metal recovery treatments from the bottom ash are the most impactful for this impact category. It is noted that the emission of NOx, especially during the electrolysis process, is the main contributor to these emissions. The FRELP process applied in [28,30], which assessed the environmental impacts linked to the disposal of 1000 kg c-Si

PV waste, led to net negative emissions of −3.1 and −16 kg NMVOC-eq, respectively as shown in Figure 19, indicating that PV recycling can provide environmental benefits.

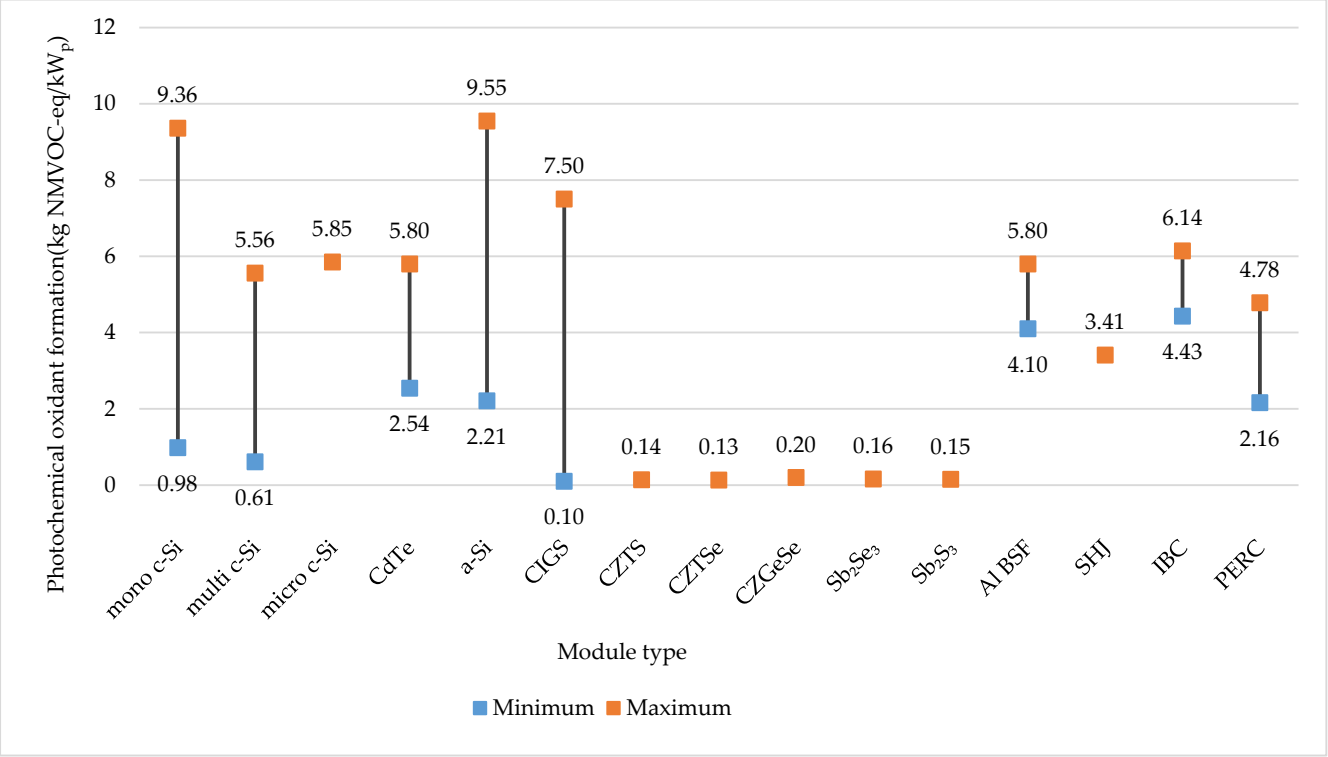


Figure 18. Analysis of PV system POFP in regard to previous studies (2015–2025).

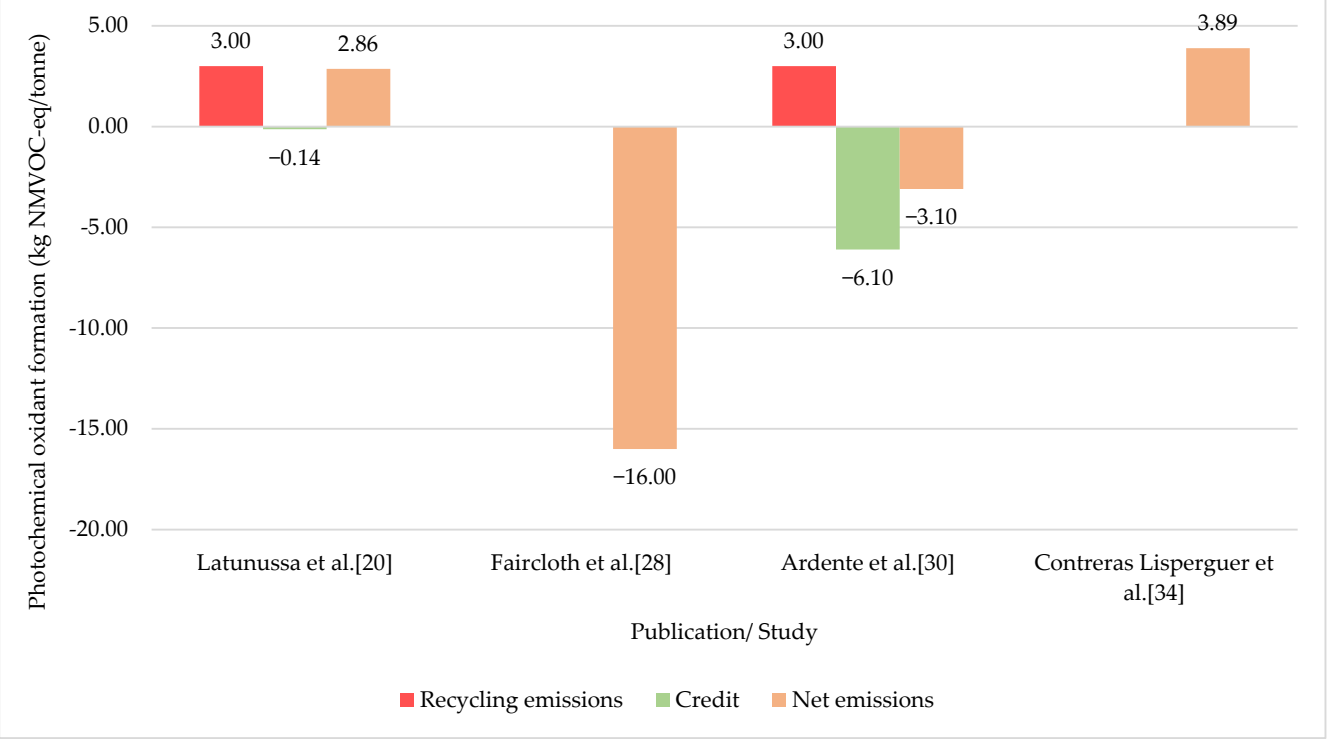


Figure 19. Analysis of PV system POFP recycling emissions [20,28,30,34].

3.10. Ozone Depletion Potential (ODP)

Ozone depletion potential quantifies the product's contribution to the thinning of the protective stratospheric ozone layer. It is typically quantified in CFC-11 equivalent (CFC-11-eq). The ozone depletion potential of various types of solar modules and cells reported in studies between 2015 and 2025 generally fall within the range of 1.85×10^{-5}–7.68×10^{-4} kg CFC-11-eq/kW_p for the different module technologies as presented in Figure 20. An exception is noted in the study by Resalati et al. [49] on different chalcogenide thin-film solar cells, which reports considerably lower values of between 6.78×10^{-10} and 1.77×10^{-9} kg CFC-11-eq/kW_p. The ozone depletion potential is generally low for all PV technologies, with most contributions arising from upstream power generation and chemical production, such as the use of chlorinated substances, as highlighted by Barnosell and Pozo [68]. These findings suggest that although minor differences exist between PV technologies, they generally pose minimal risk in terms of ozone depletion.

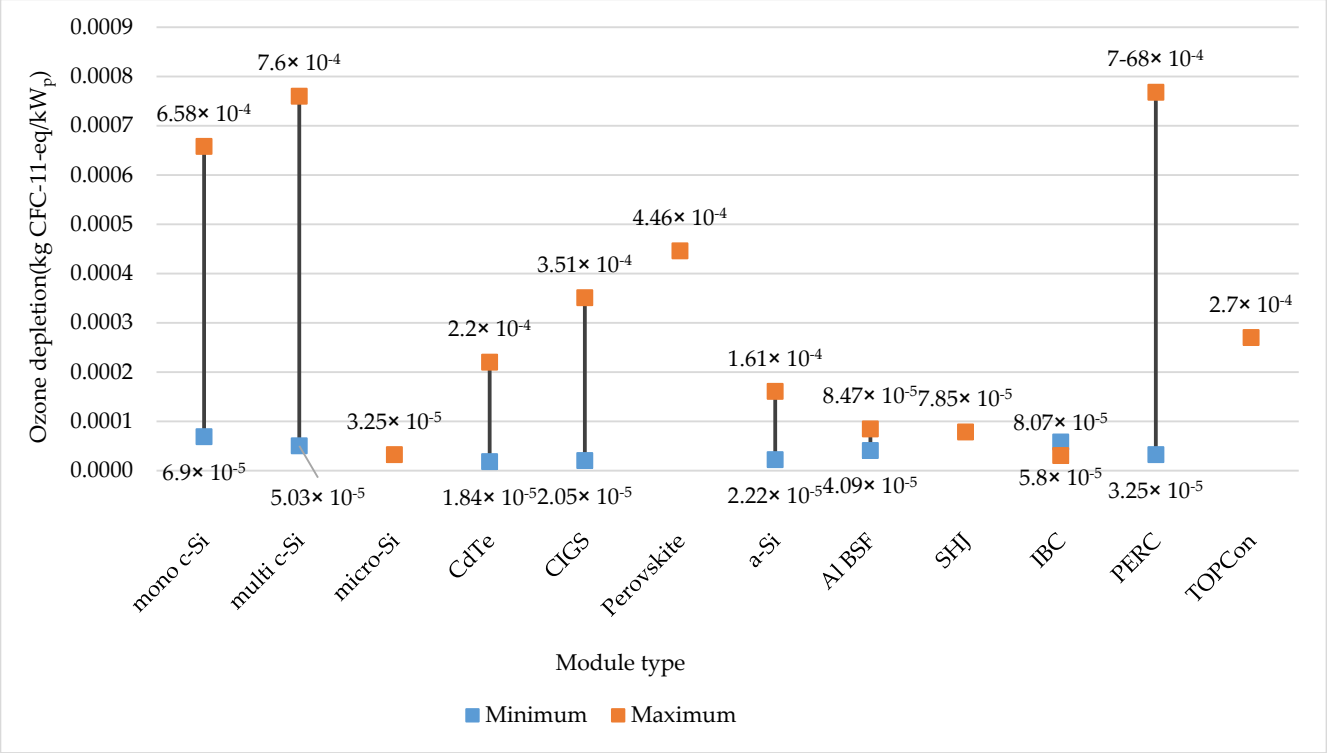


Figure 20. Analysis of PV system ODP in regard to previous studies (2015–2025).

According to Mazzi et al. [51], most of the impacts in the ozone depletion category are attributed to hydrometallurgical treatment processes. The FRELP process applied by [28,30,34,51] analyzed the environmental impacts associated with the treatment of 1000 kg c-Si PV waste, resulting in net negative emissions of -5.9×10^{-4}, -1.9×10^{-4}, -2.31×10^{-4}, and -4.4×10^{-5} kg CFC-11-eq, respectively as shown in Figure 21; thus, PV recycling provides an overall environmental benefit.

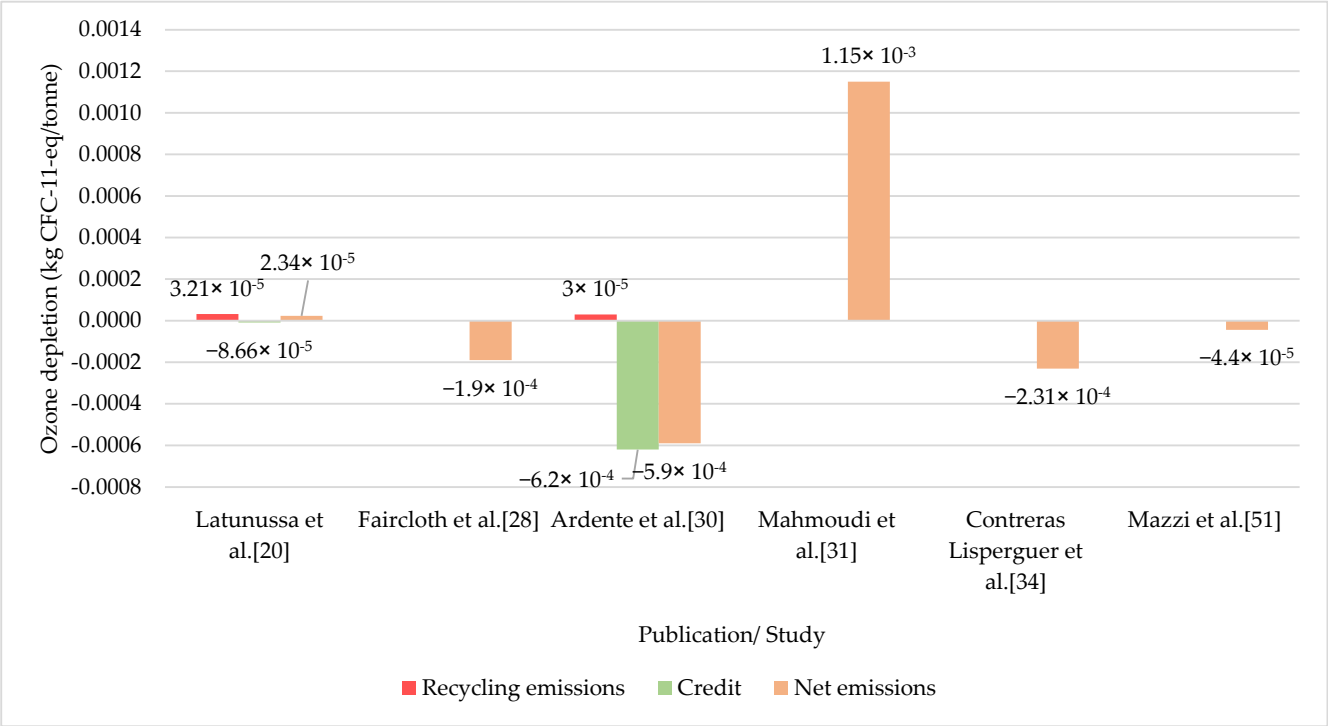


Figure 21. Analysis of PV system ODP recycling emissions [20,28,30,31,34,51].

4. Conclusions

This review analyzed the life cycle environmental impacts of various solar module and cell technologies across multiple environmental impact categories, including climate change, freshwater and marine eutrophication, acidification potential, fossil depletion, metal depletion, human toxicity, freshwater, terrestrial and marine ecotoxicity, photochemical oxidant formation, ozone depletion, particulate matter formation, ionizing radiation, land use, and water depletion. Unlike previous reviews, which mainly focused on climate change-related, energy-related, and material-based indicators, this study provides a more comprehensive review that also includes studies assessing the incorporation of recycling credits in end-of-life PV waste. It is observed that the high energy consumption associated with silicon purification and crystal growth during the manufacturing process is a major contributor to the high emissions for silicon technologies in some of the key impact categories, such as climate change, eutrophication, and acidification. Conversely, the reduced energy requirements during module production are responsible for the lower emissions of thin-film technologies and third-generation modules such as PERC and TOPCon cells, reflecting improved environmental performance. Moreover, for most of the indicators, the results from recent studies (2024) show environmental improvements for the various module and cell types, largely due to advancements in material efficiency during the optimized manufacturing process.

Concerning the environmental impacts of PV recycling, the processes involved in metal recovery, such as sieving, acid leaching, electrolysis, and acid neutralization of bottom ash, are identified as the most significant contributors to several impact categories. However, with innovative and advanced recycling methods, such as FRELP, the high recovery rates for aluminum, copper, glass, silicon, and silver effectively offset the environmental burdens of primary material production, resulting in net negative emissions across most impact categories. These results emphasize the importance of eco-design in minimizing energy consumption and reducing emissions related to most indicators over the life cycle of PV systems, especially during the manufacturing stage and end-of-life recycling of cells and

other materials. It is observed that due to the diversity of methodologies, functional units, and system boundaries across studies, there is a wide range of emission values for the various solar PV technologies, with some studies reporting very high or very low figures. This notwithstanding, it can be generally expected that with the same system boundaries, functional units, and LCA methodologies, emerging solar cells would exhibit lower emissions than conventional silicon-based solar cells.

For certain impact categories, such as human toxicity, several studies report high environmental impacts due to the emissions of certain heavy metals into the air, like arsenic, nickel, chromium, and cadmium, which are toxic and may pose serious human health risks. Additionally, the use of hazardous chemicals for silicon recovery and the incineration of EVA and plastics significantly increase toxic impacts for silicon solar cells. The high impacts on freshwater ecotoxicity are driven by the long-term release of toxic elements into freshwater resources. Similarly, indicators such as photochemical oxidant formation and particulate matter formation potential are largely influenced by metal recovery treatments from bottom ash. The rest of the impact categories, including fossil and metal depletion, ozone depletion, ionizing radiation, land use, and water depletion, generally show consistent results across studies despite being examined less extensively. This review excludes energy-related indicators such as cumulative energy demand (CED) and energy payback time (EPBT) as they have already been comprehensively addressed in previous studies and reviews. A major limitation of this study is the omission of service lifetime considerations, as most sustainability assessments are ideally based on lifetime energy generation (kWh). However, given the assumption of comparable lifetimes among the assessed module types, the use of kW_p-based emissions remains suitable for relative comparison, although possible differences in long-term performance may not be fully captured. This work is expected to support the future eco-design strategies for "designed for recycling" solar technologies, supporting the development of more circular and environmentally optimized PV systems.

Supplementary Materials: The following supporting information can be downloaded at: https://www.mdpi.com/article/10.3390/su18104639/s1, Table S1: PRISMA 2020 Checklist.

Author Contributions: conceptualization, R.B.; methodology, E.S. and R.B.; software, E.S.; validation, E.S. and R.B.; formal analysis, E.S.; investigation, E.S.; writing—original draft preparation, R.B. and E.S.; writing—review and editing, E.S. and R.B.; visualization, E.S. All authors have read and agreed to the published version of the manuscript.

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