

Review

Navigating Environmental Concerns: Assessing the Ecological Footprint of Photovoltaic-Produced Energy

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Abstract: The European Union's Green Deal concept prioritizes the installation of photovoltaic and wind turbine systems, with the aim of significantly reducing greenhouse gas emissions and expanding the use of renewable energy. The inclusion of metals/metaloids such as Cd, Pb, Ni, and As to PV panels may be a matter of concern because they may provoke numerous negative environmental effects, especially after decommissioning. Although the release of Pb and Cd from solar panels is generally low, these releases may increase, posing long-term harm. Cd and Pb, if only released from solar panels, can enter the environment, including soil and water, posing a significant risk to human health and ecosystems. Cd, in particular, can have profound and lasting negative impacts on animals and humans, affecting cellular responses, enzyme operations, and immune system functionality. Pb exposure, in turn, can induce oxidative stress and neurotoxicity, disrupt ion regulatory pathways, and impair immune function. Despite efforts to reduce the release of toxic metals from PV panels, controlling their disposal and avoiding environmental contamination remains challenging. Discovering substitute materials for PV panel manufacture, implementing enhanced recycling procedures, performing bioremediation, and enforcing stronger restrictions are among the strategies to mitigate environmental concerns.

Keywords: cadmium; lead; leaching; thin-film solar panel; perovskites; adverse outcomes; fish; mollusks; human; mitigation strategies



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1. Introduction

To achieve the 1.5 °C climate milestone, the European Union decided to implement the Green Deal paradigm [1] and do everything possible to reduce greenhouse gas emissions in the transport, electricity, and industry sectors, all of which heavily depend on fossil fuels. The valuable alternative to fossil fuels is renewable energy, sourced from solar power, wind, water, and biomass. To meet the 1.5 °C target, wind and solar need to provide 40% of the world's power by 2030 and close to 70% by 2050, despite currently supplying only 10% of the world's electricity [2]. In line with this objective, the Green Deal and REPowerEU initiatives highlight the necessity of increasing the installation of photovoltaic (PV) and wind turbine in order to utilize the capacity of solar and wind energy [1,3]. In particular, it is projected to reach 600 GW PV by 2030, even though the total installed PV capacity in the EU almost doubled from 164.19 GW in 2021 to 259.99 GW by 2023 [4].

Although PV technology seems to be a valuable replacement for fossil fuels, concerns remain regarding its environmental impact across various stages of the panel's lifespan, from raw-material mining and refining to the post-decommissioning phase. By 2030, the EU is expected to generate around 35.5 thousand tons of waste from crystalline modules, alongside an additional 1.9 thousand tons from thin-film modules [5]. The situation in the USA is even worse, with an expected one million tons of solar panel waste by 2030 [6]. Despite some researchers arguing that such waste volumes are negligible when compared to municipal waste [7], others highlight Europe's lack of readiness for this challenge [8], which could potentially retard decarbonization efforts.

Solar cells can be classified into two main categories, crystalline and thin-film solar cells, based on their device structure and architecture. Crystalline silicon (c-Si) solar cells represent the first generation of solar panel technology and are predominantly used in wafer-based solar cells, further divided into monocrystalline and polycrystalline types. Monocrystalline silicon (Mono-Si) cells, made from single-crystal silicon, exhibit enhanced efficiency levels (reaching up to 20–25%) and demonstrate improved efficiency in low-light conditions. Polycrystalline silicon (Poly-Si) cells, also known as multicrystalline cells, are produced by melting silicon crystals together. They have a slightly lower efficiency (around 15–20%) compared to mono-Si cells. However, they are generally more cost-effective to produce, which makes them a popular option for residential and commercial use. Crystalline silicon cells are the most prevalent in the solar market because they have demonstrated a high performance, long-term dependability, and well-established manufacturing procedures [9–11]. Thin-film solar cells are categorized into silicon-based and non-silicon-based technologies, representing the second and third generations of solar technology. Amorphous Silicon (a-Si) and Micromorph Silicon (a-Si/c-Si) belong to the silicon-composed thin-film solar cells. Despite their lower efficiency, often ranging from 6 to 12%, thin-film solar cells provide the advantage of using less material and consuming less energy during production, resulting in cost reduction. Non-silicon thin-film technologies include perovskite solar cells, Cadmium Telluride (CdTe), Copper Indium Gallium Selenide (CIGS), and Copper Zinc Tin Sulfide (CZTS). Perovskite solar cells, in particular, have attracted significant attention in the solar industry due to their high efficiency and potential for cost-effective manufacturing [9–11]. Considering the chemical composition of crystalline and thin-film panels, such as CdTe, $\text{CH}_3\text{NH}_3\text{PbI}_3$, CuInSe_2 , and CuGaSe_2 , can potentially exacerbate the prediction of adverse environmental outcomes for PV installations, especially those that are in the process of being decommissioned. Not by accident, the Waste Electrical and Electronic Equipment (WEEE) Directive of the European Union lists a number of metals and metalloids as PV components namely cadmium (Cd) and lead (Pb) [12], and this is noteworthy because it indicates their importance in terms of environmental regulation. When released from PV panels, Cd or Pb have the potential to contaminate soil and water, resulting in the manifestation of toxic signs that can adversely affect both humans and ecosystems.

Nevertheless, there is a scarcity of research papers that specifically focus on estimating the potential risk of Cd and Pb release, especially in terms of their bioavailability to plants and animals. To our knowledge, only a few studies have investigated the toxic impacts of released Pb, with some focusing on the determination of Pb bioavailability to plants [13] and Nemerow Contamination Index to predict soil pollution [14]. These studies have underscored the higher bioavailability of Pb to plants compared to conventional pollutants [13], as well as the significant morphological abnormalities observed in zebrafish embryos [15]. Therefore, this paper aims to analyze the possibility of Cd and Pb release from solar panels and their potential hazardous effects on animals and humans. Moreover, strategies to mitigate the release of toxic metals and minimize their adverse impacts are discussed.

2. Cadmium and Lead Emissions from Various Types of Photovoltaic Panels

Photovoltaic technology holds promise as a sustainable energy solution, yet the life cycle phases and chemical composition of crystalline and thin film panels (e.g., CdTe, CuInSe_2 , and CuGaSe_2) raise environmental concerns. Cd and Pb, components of thin-film panels and crystalline silicon solar module, are listed in the WEEE Directive [12], highlighting the need to prevent their entry into the environment.

2.1. Cd Emission

CdTe PV systems can release Cd both directly and indirectly [16]. Direct emissions of Cd occur during mining, smelting, and purification processes in PV production and indirect emissions encompass various activities throughout the production and life cycle of PV modules, including the use of fossil fuels [16]. The separation and purification of Cd

from mining waste are critical processes for CdTe manufacturing, but they are responsible for 0.015 g Cd/GWh [16] (Table 1). While emissions during accidental releases (e.g., fires) are typically minimal, they could contribute up to 0.02 g/GWh.

When considering the Cd emission per GWh of energy generated, it may initially appear low. However, factoring in the approximate mass fraction of Cd (approximately 0.1% of the CdTe absorber layer [17]) and its average mass in solar panel as high as 20 g per 2.5 m² panel [18], it becomes evident that this at first glance minor emission can have significant implications. By extrapolating these data to 2050, alongside projections of the disposal of end-of-life photovoltaic panels, which are estimated to reach 9.57 million tons [19], an anticipated installed capacity of 8519 GW [20], and current global ~5% (up to 30% in some countries, e.g., USA) market share of thin-film panel [19–21], the potential environmental impact becomes clearer. Given these projections, it is feasible to estimate that at least 340.76 tons of Cd could enter the environment in 2050 associated Cd emissions from their use, and additionally 1020.8 tons by 2050 due to the disposal of end-of-life photovoltaic panels. This aligns with the order of Cd emissions from PV calculated by Nover et al. (2017) [22]. According to their calculations, 600 GWp of installed module power in 2019 might be responsible for 2000 tons of Cd [22]. Considering that the annual emission of Cd in the EU in 2017 amounted to 24 tons [21], it becomes evident that the environmental impact of CdTe PV systems extends beyond mediate emissions.

The outcomes of leaching tests shed light on to the potential environmental impact of decommissioned CdTe PV panels when disposed of in municipal solid waste landfills [23]. Contrary to previous assumptions regarding the low solubility and bioavailability of CdTe [24,25], the outcomes of leaching tests utilizing shredded film and crushed glass reveal significant Cd leaching, up to 73%, particularly in acidic landfill conditions [23]. The significant release of Cd, exceeding regulatory thresholds, e.g., 3.24-fold higher than the Toxicity Characteristic Leaching Procedure limit of 1 mg L^{−1}, may reveal groundwater contamination and potential disruption to critical microbial communities that are able to attenuate metal pollution. Similar findings were disclosed in a long-lasting 360-day experiment: around 62% Cd released under low pH from CdTe solar panel [22]. It is highly likely that a reassessment of the disposal practices for CdTe PV panels and tactic activities to reduce their presence in municipal solid waste landfills is highly needed. Although there is a feasible risk, numerous countries have not yet introduced regulations to prevent the disposal of CdTe solar cells in municipal solid waste landfills.

While CdTe-PV panels have been implicated in environmental Cd breakthroughs, they actually release the least amounts of metals and metalloids into the environment compared with conventional modes of energy production. It is worth mentioning that, during coal combustion in power plants, the air emits up to 7.47 g Cd per GWh electricity, as can be calculated from the data presented in [26,27]. In turn, every GWh electricity generated by CdTe PV module can prevent around 4 g of Cd air emissions if used instead of or as a supplement to the heat power plant-produced electricity grid [16]. On the other hand, for silicon panels, especially monocrystalline Si panels, which currently hold a dominant position in PV panel implementation (~95%) [10,28], the anticipated indirect Cd emissions could be even higher. This projection is based on data indicating that both mono-Si and multi-Si production result in two- to three-times-higher Cd emissions compared to CdTe PV modules [29]. Also, it has been demonstrated that, throughout their life cycle, silicon panels emit, in addition to Cd, two-to-three times more Ni, Pb, and As into the atmosphere compared to CdTe-PV panels [16].

Sengül and Theis (2011) [29] corroborated these findings, but from a broader point of view. They highlighted that, not only do silicon panels release higher amounts of metals, but quantum dot PV modules, in particular, exhibit the highest emissions among all types of PV panels. In fact, the contribution of quantum dot PV modules (excluding balance of system and frames) to metal emissions ranges from 20% to 40% for various metals; however, they boast the lowest metal emissions among all other energy sources, including nuclear, wind, and hydropower [29]. It is noteworthy that thin-film panels currently comprise only

5% of the installed PV capacities worldwide and 28% in the USA [10,30], whereas crystalline silicon panel technology continues to dominate the market. However, considering the advantages of thin-film PV and the decrease in price, the thin-film photovoltaic market is projected to grow from USD 5.3 billion in 2023 to USD 11.2 billion by 2028, registering the highest compound annual growth rate (CAGR) in the forecast period of 16.0% [31]. Conversely, the Global Crystalline Silicon PV Market size is also expected to expand, reaching a staggering USD 371.74 Bn by the year 2029 with a CAGR of 12.50% [32]. This growth trajectory indicates a compelling demand for both types of PV technology and more promising future for thin-film PV. To this end, in August 2022, the US Department of Energy invested USD 20 million in the R&D of CdTe-thin-film PV modules via the newly established CdTe Accelerator Consortium [33].

To sum up, while CdTe PV systems present direct and indirect Cd emissions throughout their lifecycle, they still emit less Cd and other metals compared to silicon-based PV panels and especially compared to conventional fossil fuel installations. The potential environmental impact of Cd from CdTe PV panels might be significant, especially considering future projections for their disposal. This highlights the need for strategic activities, such as encapsulation and finding alternative materials that can at least partially replace Cd, to mitigate these environmental concerns.

2.2. Lead Emission

Not only does Cd have the potential to be released from solar panels, but other toxic elements may also be released, posing a threat to the environment and human health (Table 1). High emissions of Pb from various types of solar panels have been observed, ranging from 7 to 20 g/GWh [16]. This emphasizes a greater overall concern beyond Cd alone, especially when considering the WHO's recommended tolerable weekly intake values for Pb [34], which stand at 25 µg/kg body weight.

Perovskite solar cells (PSCs) hold great promise, with high efficiencies, low manufacturing costs (about 38.69 USD/m² compared to 62.90–79.31 USD/m² for crystalline PV), and a similar lifespan to common crystalline panels. Most PSC formulations contain Pb, making them ideal for researching phonon confinement. The Pb 6s orbitals contribute to the highly tunable bandgap, high carrier mobilities, low exciton binding energies, and strong multiple exciton generation effects [35]. Nevertheless, with a relatively high Pb content, some perovskite formulations may contain up to 10% Pb. Moreover, analyses of the total organic carbon and chemical oxygen demand suggested that discarded PSCs might elevate oxygen consumption and release CO₂ into the environment [36], raising significant health and environmental concerns.

According to Wu et al. (2022) [37], perovskite modules will likely contain 0% to 0.689% Pb, and using the molecular weight and the densities of common perovskites, it is estimated to be as high as 4.09 g/cm³ for MAPbI₃, 3.83 g/cm³ for MAPbBr₃, 4.10 g/cm³ for FAPbI₃, and 5.39 g/cm³ for CsPbI₃. It has been shown that the cumulative leaching concentration of Pb from PSCs exceeded 5.0 mg/L, designating discarded PSCs as hazardous waste [36]. Hailegnaw et al. (2015) [38] and Li et al. (2020) [13] found approximately 0.4 g/m² and 0.8 g/m² of Pb in the perovskite layer, assuming an absorber layer thickness of 300 nm and 600 nm, correspondingly, with total Pb intensity showing content per unit electricity produced of 38 µg/kWh [39]. If all of this Pb from 300 nm and 600 nm perovskite layers is released into the top 1 cm of typical soil, the concentration would increase by about 70 ppm [38], reaching 4.0 mg/kg [13], correspondingly. This represents a significant increase when considering the average concentration of Pb in soils, which generally falls within the range from 15 to 40 ppm [40].

On the global scale, with an efficiency of approximately 25%, perovskite solar panels contribute to 1 GW of solar PV capacity, containing around 3.5 tons of Pb within a 500 nm thick Pb-based perovskite layer [41]. When we take into account the projected PV perovskite installations, potentially reaching 100 GW by 2030 compared to 2 GW in 2026 [42], and the degradation rate of around 1% per annum [43], the possible emissions of Pb might

be 3.5 tons of Pb in 2030 compared to 0.07 tons of Pb in 2026 in the best-case scenario. However, in the worst-case scenario, which might occur with a 5.3% chance of material release from dumped modules [44], where up to 17.6% Pb ought to be released from the solar cell [45], 616 tons and 1.21 tons of Pb are expected in 2030 and 2026, respectively. These numbers are even higher than the annual Pb emissions of the United Kingdom in 2018, which were stated to be as high as 93 metric tons [46], and align with the expected volumes of Pb that could enter the environment from the disposal of PV waste in Italy. Specifically, it is expected that approximately 2900 tons of Pb could potentially be lost due to improper disposal of PV waste in Italy [47]. Meanwhile, Billen et al. (2019) performed a life cycle assessment in order to evaluate the emissions of Pb and its potential for harm to humans [48]. The findings indicated that if PSCs achieved a capacity of 2400 GWp, the perovskite light harvester would require just 17,000 metric tons of Pb, which is equivalent to about 1.1% of the annual Pb usage of 1.6 million metric tons in the US [48].

Table 1. Cadmium and lead content in PV cells and emission-related parameters.

	Cd	Ref	Pb	Ref
Concentration	0.1% of the CdTe absorber layer $8 \text{ g} \cdot \text{m}^{-2}$ $78 \text{ mg} \cdot \text{W}^{-1}$ (0.4 μm)	[17] [18] [22]	MAPbI ₃ — $4.09 \text{ g} \cdot \text{cm}^{-3}$	[37]
			MAPbBr ₃ — $3.83 \text{ g} \cdot \text{cm}^{-3}$	[37]
			FAPbI ₃ — $4.10 \text{ g} \cdot \text{cm}^{-3}$	[37]
			CsPbI ₃ — $5.39 \text{ g} \cdot \text{cm}^{-3}$	[37]
			Perovskite (0.3 μm)— $0.4 \text{ g} \cdot \text{m}^{-2}$	[38]
			Perovskite (0.6 μm)— $0.8 \text{ g} \cdot \text{m}^{-2}$	[13]
			c-Si— $6.1 \text{ g} \cdot \text{m}^{-2}$	[37]
Direct emission	$0.02 \text{ g} \cdot \text{GWh}^{-1}$ due to accidental release	[16]	7 to 20 $\text{g} \cdot \text{GWh}^{-1}$	[16]
	$76.66 \text{ tons} \cdot \text{GW}^{-1} \cdot \text{y}^{-1}$	[22]		
Indirect emission	$0.015 \text{ g Cd} \cdot \text{GWh}^{-1}$	[16]	ND	
Metal intensity, per U electricity produced	ND		$38 \mu\text{g} \cdot \text{kWh}^{-1}$	[39]
Diffusion coefficient	$3 \times 10^{-17} \text{ cm}^2 \cdot \text{s}^{-1}$ (pH 4)	[22]	$3 \times 10^{-15} \text{ m}^2 \cdot \text{s}^{-1}$	[49]
Leaching/leakage rate	Up to 73% (pH < 7)	[23]	$>5 \text{ mg} \cdot \text{L}^{-1}$	[36]
	62% (pH < 7)	[22]	$30 \text{ mg h}^{-1} \text{m}^{-2}$	[50]

When Pb emission from perovskite thin film is expected, because of Pb in perovskite formulation, not many researchers expect Pb emissions from commonly used crystalline silicon PV. However, the interconnect within these modules is typically made using lead-/tin-coated copper ribbons, connecting the front-side busbar of one cell to the backside contact on the adjacent cell in that substring of the module [51]. Although the mass fraction of Pb in the interconnect is believed to be not higher than 0.053% [47], the amount of Pb used in commercial c-Si modules might be quite significant. It is estimated that the current Pb content in 60-cell c-Si panels is around 576 mg/kg [47], or 6.1 g/m^2 [37]. Considering that crystalline silicon represents 95% of the market size [28], this implies that a significant amount of Pb should enter the environment.

Taking into account the projected 2030 and 2050 PV installations of 2840 GW and 8519 GW, respectively [10], along with the market share of crystalline panels [28], average panel efficiency [52], and a calculated lead content as high as 6.1 g/m^2 [37], the Pb content in crystalline panels could reach approximately 779.6 and 2626.97 tons by 2030 and 2050, respectively. Assuming an average degradation rate of around 1% per annum [43], there could be emissions of 7.8 and 26.3 tons per year. Such emission volumes are highly likely to be far from the lead-zero policy presented in the “Directive on the restriction of the use of certain hazardous substances in electrical and electronic equipment” [53].

To sum up, while Pb emissions from solar panels are relatively low compared to global lead emissions, their introduction into the environment can lead to soil and water enrichment with Pb, which can have adverse long-term effects on wildlife and humans. The issue is pressing due to the large amount of PV waste expected in the near future.

3. Potential Adverse Outcomes of Cadmium and Lead Exposure in Living Organisms

The release of Cd or Pb from solar panels presents a potential hazard to the environment by contaminating soil and water. Despite limited information on the projected volumes of Cd and Pb released, when considering existing the projections on PV installations, anticipated solar panel waste, average degradation rates, and associated calculations presented above, it becomes evident that they pose a risk to plants, animals, and humans, and their intensity is going to increase. However, research assessing the effects of Cd and Pb released from solar panels on biota is scarce. Among the few studies conducted, it has been demonstrated that lead leaking from perovskite can enter plants ten times more effectively than other lead contaminants resulting from human activities, leading to adverse effects, such as blackening and rotting in mint plants (*Mentha spicata*), indicative of Pb intoxication even at levels of PSCs of 250 mg kg^{-1} that are considered safe by agricultural regulations in China [13].

Once Cd enters living organisms, it can have profound and lasting negative impacts on animals, humans, and ecosystems. Cd can enter organisms like fish and mollusks by using various transport routes, such as the gill epithelia or intestinal tract, facilitated by transporters like ZIP14, DMT1, and ATP7A [54]. Entered Cd can initiate diverse cellular responses even at low concentrations [55]. The detrimental outcomes of Cd exposure encompass the disruption of calcium and sodium homeostasis; the formation of nuclear anomalies like micronuclei, vacuolated nuclei, and irregular nucleus shapes in the hemocytes and erythrocytes of mollusks and fish [56]; and morphological and functional disturbances in organelles like mitochondria [57] and lysosomes [58] (Figure 1). Cd exposure leads to an increase in reactive oxygen species, triggering oxidative stress via disturbances in the electron transport chain or promoting the release of free iron in the Fenton reaction [59]. Moreover, this stress is exacerbated by Cd disrupting enzyme operations in the electron transport system, resulting in reduced ATP production, the inhibition of substrate oxidation, and increased mitochondrial ROS production [57,60].

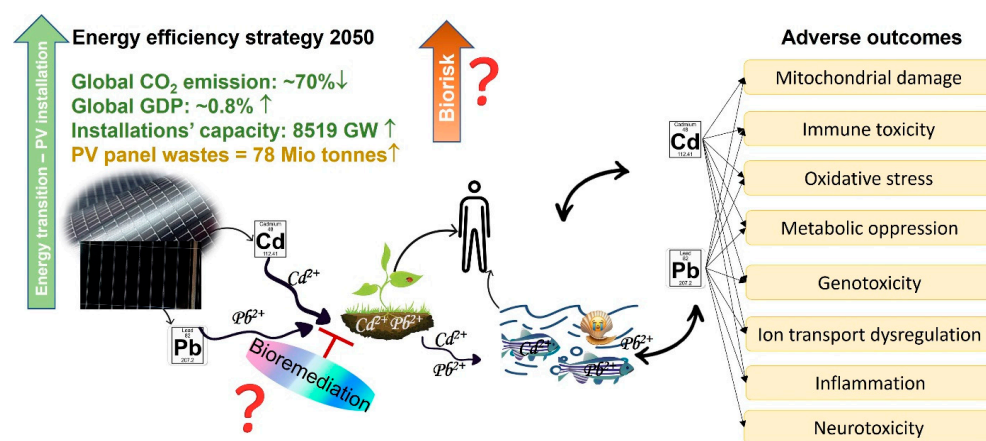


Figure 1. Potential release pathways of cadmium and lead from solar panels into the environment, including their environmental entry points, targets, and associated adverse effects. Cadmium and lead, released directly or indirectly from solar panels, can penetrate the environment. They may accumulate in soils, subsequently entering plants, or contaminate ground and surface water, further possibly accumulating in aquatic animals. As humans are at the top of the trophic chain, they may be affected by consuming exposed food, including vegetables, seafood, and freshwater fish. Implementation of bioremediation techniques can help reduce the potential negative impacts of released xenobiotics on the environment and living organisms. Numerical data come from www.irena.org (accessed on 19 June 2024), iea-pvps.org (accessed on 19 June 2024), ise.fraunhofer.de (accessed on 19 June 2024).

Cd can affect the immune system of aquatic biota; it predominantly influences the non-specific immune response, leading to a reduction in hemocytes and white blood cell count,

lymphocyte proliferation, affection of phagocytic activity estimated as acid phosphatase activity of lysosomes and neutrophil phagocytosis rate, and macrophage activity [61]. Moreover, Cd accumulation induced oxidative stress and inflammation-related gene expression, while inhibiting neutrophil extracellular traps' release and respiratory burst, as shown for common carp [62]. Cd exposure targets the immune system, compromising its structural integrity and functionality, resulting in immunosuppression [62]. In vitro and in vivo studies on *Ictalurus melas* have demonstrated significant inhibition of lymphocyte proliferation at Cd concentrations exceeding 2 $\mu\text{mol/L}$ [63]. High Cd concentrations inhibit the activities of lysozyme, alkaline phosphatase, and acid phosphatase in the liver of yellow catfish and blue mussels [64]. Also, alongside a linear increase in Cd accumulation in blue mussel tissues after the action of Cd ranging from 200 $\mu\text{g/L}$ to 600 $\mu\text{g/L}$, prominent immune disorders without significant physiological disturbances were observed [65]. Additionally, Cd induces neutrophil proliferation in zebrafish larvae, causing massive inflammation [66]. It is highly likely that the accumulation of Cd can trigger molecular and cellular dysfunctions much earlier than physiological disorders. In the case of using only physiological biomarkers or application of bioindication approaches, it is possible to underestimate Cd's adverse effects on organisms and anticipate organismal adverse outcomes in long-term projections.

The detrimental impacts of Cd contamination in aquatic life extend to humans as well. It has been evident that Cd causes immunological reactions, interferes with immune cell function, and causes a number of health issues. The immunotoxic nature of Cd is proven by its disruption of T-lymphocyte subsets and antibody generation, alterations in cytokine release, activation of oxidative stress, and modulation of immune cell activity and apoptosis [67,68]. Furthermore, Cd can cause mitochondrial malfunction, specifically affecting electron transport chain complexes. Enzyme inhibition, elevated generation of reactive oxygen species, and disturbance of membrane potential are signs of mitochondrial dysfunction. For instance, Cd-induced mitochondrial dysfunction in HEK293 cells includes increased permeability, respiratory inhibition, and oxidative-stress induction [69]. These findings emphasize the necessity for further research in order to understand the mechanisms of Cd toxicity, particularly regarding its potential release from PV panels and its subsequent bioavailability to animal models. Such research is not only important for the prediction of potential adverse outcomes for both biota and humans but also supports findings related to appropriate strategies to mitigate the harmful effects of Cd contamination.

Accumulation of Pb in animal tissues triggers the generation of reactive oxygen species, leading to a substantial increase in lipid peroxidation and a decline in total antioxidant capacity and glutathione levels, thus inducing oxidative stress and provokes DNA damage [70]. Moreover, Pb is able to disrupt ion regulatory pathways in water animals, resulting in hypocalcemia, perturbed sodium influx rates, and an imbalance in the electrochemical gradient across cell membranes, thus exacerbating cellular dysfunction [71,72]. Lead exposure also influences cytokine expression and mitogen-activated protein kinase (MAPK) activity, which modulates immune responses. Cytokines such as $\text{TNF-}\alpha$ and Interleukin 10, which play key roles for immune system regulation and inflammation, can be also affected. Pb-induced MAPK activation can lead to apoptosis and stress reactions, further impairing immune function [72]. Additionally, Pb can negatively impact growth performance; erythrocyte morphology; and the histomorphology of the gills, liver, and intestine in both juvenile and mature fish specimens [71,73].

Pb exposure can pose a significant risk to human health (Figure 1). People can be affected both directly and indirectly through food consumption. Numerous studies have demonstrated its detrimental effects on physiological processes, particularly within the nervous system, where it can induce synaptic damage, neurodegenerative diseases, and diminished neurotransmission [74]. Lead exposure triggers depolarization of mitochondrial membrane potential and a reduction in mitochondrial mass, ATP levels, and mtDNA copy number. Similar to its effects on aquatic animals, lead suppresses the functional capabilities of the electron transport chain (ETC), leading to decreased expression of mitochondrial ETC

complex proteins (e.g., ATP5A and COXIV) and downregulation of ETC complex gene expressions such as COXIV and ATP5F1 [75]. This disruption also affects ionic processes and induces oxidative stress, resulting in enzyme and protein dysfunctions. These disturbances can cause ATP depletion, metabolic alterations, and disturbances in neurotransmitter systems, consequently affecting cognitive and brain function. Additionally, Pb toxicity in erythrocytes manifests through mechanisms such as compositional phospholipid distortion, transmethylation inhibition, and exacerbated phospholipid peroxidative damage [76].

In conclusion, there is a serious risk to the environment, ecosystems, and public health from the discharge of Cd and Pb from solar panels. Despite there being limited data on projected volumes of Cd and Pb emission and leachates, existing projections on PV installations and anticipated solar panel waste, alongside average degradation rates, indicate a growing risk to plants, animals, and humans. Water animals, particularly fish, could be used as an alternate model for research in order to fully estimate the bioavailability of these metals to aquatic and wildlife species. Further research into the mechanisms of PV-originated Pb and Cd toxicity is also necessary in order to facilitate the extrapolation of findings to consequences for human health.

4. Methods to Minimize the Release of Toxic Metals from Solar Panels

It is widely established that solar panels, both halide perovskite and crystalline, can be vulnerable to degradation in outdoor conditions, such as exposure to moisture/water, ultraviolet light, damp heat, and oxygen, especially upon physical damage or failure of encapsulation. As was mentioned in the previous section, if PV modules were to break while deployed, Cd and Pb could leach into the surrounding environment, and with the increasing use of PV panels comes a growing concern for the safety of solar cells. Therefore, researchers are focusing on replacing the Pb and Cd with less toxic elements, while maintaining efficiency and improving the stability in case of the PSCs, as well as finding the suitable recycling approach to assure the fullest recycle.

4.1. Physical Encapsulation and Chemical Absorption

One of the possible practical solutions to mitigate Pb leakage involves physical encapsulation and chemical absorption [37], without significantly compromising efficiency. Encapsulation methods for laboratory-scale applications can be broadly categorized into two types: thin-film encapsulation and cover-glass encapsulation; the latter is further divided into blanket and edge encapsulation [77]. An ideal encapsulation system would incorporate several key elements: a thin passivation layer, akin to those utilized in OLED encapsulation, preferably deposited through solution processing or thermal evaporation to streamline processing and equipment transfer; an encapsulant that preserves perovskite integrity and offers strong adhesion; an edge sealant, utilizing desiccant-filled PIB with outer edge protection to safeguard the PIB; a non-permeable cover (such as glass or metal) for enhanced thermal management; and integrated lead sequestration [77]. While effective physical encapsulation can prevent accidental leaching of Cd and Pb into the environment, it adds an additional step during recycling due to the removal of ethylene-vinyl acetate (EVA). However, this step can be simplified through heat treatment and the use of organic solvents [78].

4.2. Cd and Pb-Free Alternatives and Their Challenges

4.2.1. Pb-Free Thin-Film Solar Panels

Exploring alternatives to reduce the usage and emission of toxic elements in PSCs has led to the development of Pb-free options. Among these, Sn-based, Ge-based, Bi-based, and Sb-based PSCs have shown significant performances. However, their power-conversion efficiencies currently stand at approximately 14%, 7.1%, 3.6%, and 2.8%, respectively—still considerably lower than conventional c-Si technologies [79]. While Sn might serve as a substitute for Pb due to similarity in physicochemical properties and electron configuration, Sn-based modules face stability challenges. This is because Sn ions tend to remain in

the Sn^{4+} oxidation state, which is less conducive to substitution in PSCs compared to the Sn^{2+} state found in Pb-based counterparts [80]. Moreover, tin halide perovskites degrade into SnO_2 and SnI_4 , with the SnI_4 being highly reactive with water, potentially leading to respiratory disorders and adverse effects to water animals and plants. Other metal/metalloids, such as Cu, Bi, and Ge, have also been explored, but their devices suffer from poor performance due to significant trap states and pronounced ion-migration issues [81].

4.2.2. Cd-Free Thin-Film Solar Panels

Recent research indicates promising developments in constructing Cd-free thin-film solar panels, particularly highlighting amorphous silicon and Copper Zinc Tin Sulfide (CZTS) as promising alternatives [11]. Ahmad et al. (2023) introduced Ag-CZTSSe thin-film solar cells with an eco-friendly zinc tin oxide (ZTO) buffer layer, deposited via atomic layer deposition [82]. Through the optimization of stoichiometry and thickness, they significantly enhanced the efficiency of kesterite solar cells. Experimental analyses revealed that a 10 nm ZTO layer with specific atomic layer deposition pulse ratios improved band alignment and electron transport and reduced interface defects and recombination. The wide bandgap of ZTO further minimized light loss compared to CdS, resulting in superior carrier collection and a record efficiency of 11.8% [82], which, however, lags behind conventional CdTe-PV or crystalline PV. Similarly, in Cd(S,Se,Te) thin-film solar cells, introducing a bandgap gradient through a Cd(O,S,Se,Te) region near the front junction, facilitated by oxygenated CdS and CdSe layers, enhanced open-circuit voltages. An efficiency of 20.03% with a VOC of 0.863 V in their best-performing device was achieved [83].

4.3. Recycling Technologies

In contrast to conventional disposal methods, like landfills and incineration, for end-of-life solar cells, recycling offers significant economic and ecological advantages for their final disposal. Recycling techniques are typically categorized into physical and chemical methods (Figure 2), with contemporary approaches often combining both for comprehensive material recovery [84]. Physical processes involve initial the disassembly of modules to extract the aluminum frame, followed by shredding to obtain a powder. This powder is then subjected to heavy medium separation to separate glass and metals from the polymer. Metals can be recycled, as an example, using gradient density techniques, with various densities of NaCl/Na polytungstate of 1.2 g/cm³, 1.3 g/cm³, 1.4 g/cm³, and 1.5 g/cm³ employed for effective separation [85]. Pagnanelli et al. (2017) proposed a method involving crushing and screening waste solar panels and then dividing them into fractions based on particle size [86]. Glass recovery from particles sized 0.4–1 mm was direct, while larger particles required heat treatment to remove EVA. Metals were predominantly concentrated in particles smaller than 0.4 mm and were recovered using a leaching process with sulfuric acid and hydrogen peroxide, achieving a final product recovery rate of 91% [86]. A comparable method using sulfuric acid and hydrogen peroxide effectively extracted both cadmium and tellurium from CdTe thin-film panels, followed by their separation using cation-exchange resins [87].

Another physical method involves shredding solar panels using a knife mill, followed by medium separation, milling, and sieving, or electrostatic separation. This method has shown glass recovery rates ranging from 76% to 100% at approximately 100% purity and metal recovery rates around 67% purity [84]. Despite the fact that this approach is environmentally friendly, the resulting products had lower purity, so further research is needed to enhance recovery rates and material purity. To address this, the voltage pulse method was introduced to improve metal grades [85]. Song et al. (2020) implemented a high-voltage fragmentation method, achieving high recovery rates for copper (~95%), silver (~96%), aluminum (~85%), lead (~85%), and silicon (~87%) under optimized parameters [88].

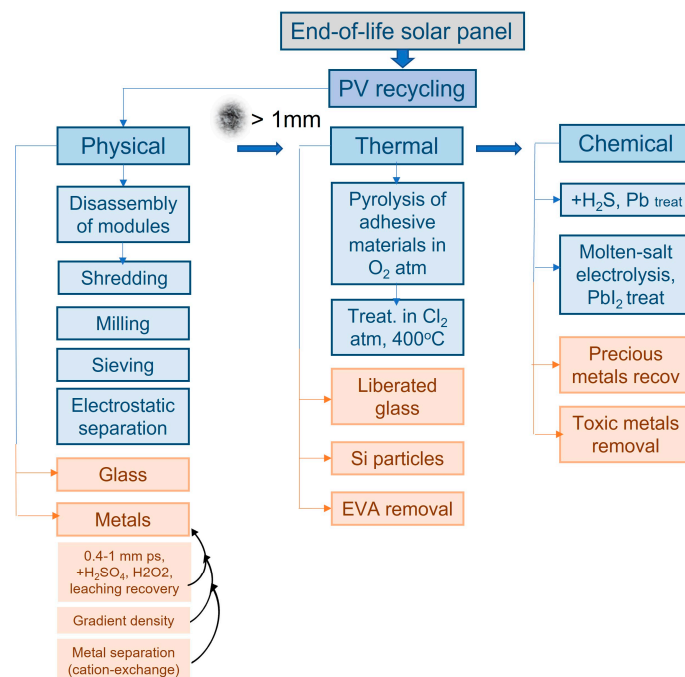


Figure 2. Recycling pathway of photovoltaic modules, with possible key steps (in blue) and resulted conserve resources (in pastel).

Chemical processes are crucial for the hydrometallurgical recovery of valuable elements, while also effectively trapping hazardous materials like Pb. An analysis of the solubility product constant of lead compounds has revealed challenges in its removal. While chloride ions can reduce lead ion concentration, achieving this requires high concentrations of HCl, at least 1.08 mol/L. Alternatively, sulfide ions can remove lead, but this method generates toxic H₂S. Given the small solubility product constant of PbSO₄, using sulfate ions for removal appears more feasible. However, maintaining a sufficiently high sulfate ion concentration—as much as 1.82×10^{-3} mol/L—is necessary for at least 81.56% lead removal [89].

Various approaches exist for the recovery of Pb from perovskite solar cells, distinct from those employed for crystalline panels. For instance, Wang et al. (2022) introduced a novel solution called molten salt electrolysis, designed to recover Pb and iodine (I₂) from spent Pb-based PSCs [90]. This method involves dissolving lead iodide (PbI₂) in molten LiCl-KCl and electrochemically converting it into Pb and I₂. The high recovery efficiency of Pb, coupled with the rapid leaching and recovery process, demonstrates molten salt electrolysis to be a simple and eco-friendly approach to eliminate Pb contamination and close the iodine cycle [90]. Moreover, Chen et al. (2021) demonstrated the feasibility of recycling lead and transparent conductors from decommissioned perovskite solar modules [41]. Through delamination and dimethylformamide extraction, lead is isolated using weakly acidic cation exchange resin, subsequently released as soluble Pb(NO₃)₂ and precipitated as PbI₂, achieving a recycling efficiency of 99.2% [67]. Schmidt et al. (2023) further contributed to Pb recycling by presenting an organic solvent-free process, specifically targeting PbI₂ [91]. Employing hot water extraction, they achieved efficient lead recovery from synthetic precursor mixes and perovskite modules, yielding high-purity PbI₂ precipitation (>95.9%). The temperature-dependent solubility of PbI₂ in water allows for straightforward two-cycle lead extraction, recovering $94.4 \pm 5.6\%$ in the first cycle and an additional $10.0 \pm 5.2\%$ in the second. The disposal of extraction residues as non-hazardous waste mitigates potential environmental risks [91].

It is worth noting that research into the recycling of solar panels has seen a shift towards mechanical methods as more environmentally friendly methods, away from traditional chemical or physicochemical processes. While the latter are efficient in extracting

toxic elements [41,89], mechanical techniques offer promising ways for improvement. A recent innovation involves the mechanical recycling process for CdTe/CdS thin-film solar cell modules. Here, modules undergo mechanical disintegration into fragments, followed by exposure to an oxygen-containing atmosphere at a minimum temperature of 300 °C, inducing pyrolysis of adhesive materials, typically hydrocarbon-based plastics. The resultant gaseous decomposition products are then discharged. Subsequently, the fragments, now free of adhesive residue, are subjected to a chlorine-containing gas atmosphere at temperatures exceeding 400 °C. This initiates an etching process wherein CdCl₂ and TeCl₄, by-products of the process, condense and precipitate upon cooling [92]. Also, the Double Green Process represents a notable advancement, emphasizing mechanical and hydromechanical treatments. It uses a few chemicals, and it is characterized by a greater level of automation and a high flexibility in production capacity [93]. Compared to conventional methods, the Double Green Process demonstrates potential for lower environmental impact, particularly related to ionizing radiation and land occupation, scalability, decentralization, automation, and enhanced safety, as evidenced by life cycle assessments [93]. The use of flotation allows for a greater percentage of recovered materials and higher revenues from an economic point of view. The next example of a low-cost and environmentally friendly recycling approach, consisting of module deframing, laminate shredding, and material concentration using electrostatic separation, was proposed by Dias et al. (2022) [94]. The economic assessment indicates that this process has the potential to be more profitable than full-recovery alternatives under several conditions: (i) at lower waste volumes (less than 4 kt/y) due to the reduced capital cost of equipment; (ii) in the absence of a market for recovered glass, as is the case in many regions; or (iii) when the end-processing industry is located at a great distance, since only the valuable mixture would need to be transported [94]. These approaches help us take a step forward in sustainability PV panel recycling.

Economically, recycling offers significant advantages over traditional waste disposal methods, especially considering the discrepancy in disposal costs between hazardous and municipal waste [91]. Compliance with regulatory standards such as Waste from Electrical and Electronic Equipment necessitates efficient harmful metals recycling to meet specified rates. Additionally, recycling mitigates the social and environmental impacts associated with PSC materials throughout their lifecycle, from production to end-of-life phases, including potential leaching concerns.

4.4. Bioremediation as a Valuable Technique to Attenuate Adverse Outcomes of Released Metals

Various methods have been utilized to extract metals from contaminated areas, such as electrolysis, ion exchange, precipitation, and reverse osmosis. However, these traditional approaches often fail to meet environmental-friendliness criteria and are not cost-effective. Bioremediation presents itself as an alternative option for removing metals like Cd and Pb from polluted soil and water [95]. Plants, in particular, possess bioremediation capabilities due to their significant sorption capacity for metals. The soil-to-plant transfer coefficient of Cd ranges from 3.9 to 3340, depending on metal availability and plant species [96]. As an example, alpine pennygrass *Noccaea caerulea* (formerly *Thlaspi aerulescens*) can accumulate up to 1800 mg·kg⁻¹ of Cd, while *Thlaspi rotundifolium* demonstrates a high ability to accumulate Pb [97]. It is worth mentioning that, despite Pb's limited mobility in soil, its extraction rate is constrained by solubility and diffusion to the root surface, making some plant species pretty successful in regard accumulating to Pb. Furthermore, the efficacy of Cd phytoremediation and Cd bioavailability can be enhanced through the use of plant growth-promoting bacteria.

Another option to remove metals from soil and water media is the implementation of microorganisms [98]. Microbial enzymes such as oxidoreductases and dehalogenases can effectively remove metals, including Cd and Pb, from agricultural soils. Plant-secreted bioactive compounds, in conjunction with plant growth-promoting rhizobacteria, can generate enzymes such as 1-aminocyclopropane-1-carboxylase deaminase, which assist in

the decomposition of metal pollutants. Additionally, the enzyme phytochelatin synthase produces phytochelatins— γ -Glu-Cys(n) Gly polypeptides—which are thiol-rich metal chelators similar to metallothioneins in animals, contributing to metal sequestration and reducing their harmful impacts [99,100]. Moreover, microbially induced precipitation, particularly carbonate precipitation, immobilizes Cd ions, thereby decreasing their mobility.

Metagenomics techniques, such as high-throughput sequencing, can help identify potential candidates for metal bioremediation by detecting microbes' enzymes, metabolites, and bioactive compounds sensitive to environmental pollution [101]. Despite these promising expectations, bioremediation techniques have limited applications in metal decontamination, which might appear during the commissioning and decommissioning of PV systems. Therefore, these techniques need to be further analyzed and refined for such applications.

At some point, ensuring the safety and sustainability of solar panels depends on effectively reducing environmental risks associated with their materials and decomposition. Researchers are currently engaged in the active development of alternatives to Pb and Cd, as well as enhancing techniques for encapsulating and improving recycling processes. These advances are designed to improve the efficiency, stability, and environmental friendliness of solar cells, ensuring a more sustainable future. Nevertheless, more progress is needed in order to tackle issues related to the performance of materials, efficient recycling process, and mitigation activities, e.g., bioremediation, thereby promoting the development of sustainable solar energy solutions.

5. Conclusions and Future Perspectives

Although solar panels have lower CO₂ emissions than fossil fuels, the anticipated increase in PV installations and panel waste pushes towards profound studies of their overall environmental impact. This includes assessing metal release potentials, possible adverse outcomes, and corresponding mitigation actions. In particular, a review of the literature using specific keywords reveals that fewer than 20 papers are dedicated to determining Cd and Pb mass fractions in solar panels and their potential release. Existing data, such as the estimated Cd release of up to 76.66 tons·GW⁻¹·y⁻¹ [22], highlight the potential risks to the environment and biota. Therefore, there is an urgent requirement for extensive research that specifically examines the bioavailability and long-term ecological consequences of toxic metals. Research should focus on clarifying the ways in which metals are released and accumulated in various environmental matrices, such as soil and/or water, niches, and living organisms.

Furthermore, the progress in PV technology should give priority to developing Cd- and Pb-free solar panels, or at least decrease their concentration to reach safe limits. This entails investigating semiconductor materials that are less hazardous but maintain comparable efficiency. For example, conducting studies on alternative materials, such as Ge, Bi, and Sb, might address the existing constraints in efficiency and stability.

Encapsulation technological advancements are crucial as well. It is important to develop encapsulating materials that are more durable and affordable and that can offer long-lasting protection against environmental deterioration. Furthermore, the implementation of comprehensive recycling and waste management procedures appears to effectively manage PV panel disposal at the end of their lifespan. This includes the development of effective recycling methodologies that can recover and repurpose valuable resources, while minimizing ecological pollution.

Bioremediation is a potentially effective method for reducing the environmental consequences of solar panel waste; however, it has not been thoroughly investigated. Future research should prioritize the identification and enhancement of plant species and microbial strains that have a strong ability to accumulate and store metals. In addition, metagenomics and other sophisticated biotechnological techniques can be used to gain a deeper understanding of and improve the processes involved in metal bioremediation.

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