

Economic and Environmental Sustainability of Copper Indium Gallium Selenide (CIGS) Solar Panels Recycling

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Abstract

End-of-life management of copper indium gallium selenide (CIGS) thin-film solar photovoltaics (PV) panels is crucial due to the necessity of recycling valuable elements such as indium (\$400/kg) and gallium (\$618/kg), ensuring both economic viability and environmental sustainability. In this study, we analyze the private and external costs of end-of-life management for CIGS PV designed for mass-scale recycling. Our findings reveal that the private and external costs of end-of-life management range from ~\$3.5 to \$4.5 and ~\$3.0 to \$4.0 per m², respectively. The chemicals utilized in the recycling process, particularly NaOH and HCl, significantly contribute to climate change, photochemical oxidant formation, particulate matter formation, and freshwater eutrophication impact categories, accounting for ~50% to 90% of the impacts. Furthermore, we found the net cost of recycling by subtracting the economic benefit obtained from recovered materials from the sum of private and external costs, revealing values ranging between \$4.3 and \$5.7 per m² of CIGS PV module. These findings suggest that there is room to reduce the net cost further by recovering more materials from the CIGS PV modules components.

Keywords: PV recycling, end-of-life of PV, CIGS PV, private cost, external cost

1. Introduction

Copper indium gallium selenide (CIGS) is a commercially available, thin-film photovoltaic (PV) technology (Kim et al., 2021), with efficiencies of 23.6% at the cell and 19.2% at the module level (NREL, 2024). As of 2023, the global installed capacity of CIGS PV has surpassed 12GW (Fraunhofer Institute of Solar Energy Systems, 2023). The active layer of CIGS PV consists of copper, indium, gallium, and selenide to harness the light deposited on the glass, metal, or plastic layer (Mohammad Bagher, 2015). CIGS PV panels offer significant advantages in terms of flexibility and application versatility compared to crystalline silicon and cadmium telluride PV technologies (Alarifi, 2023; Pagliaro et al., 2008). CIGS PV panels have a lower temperature coefficient, which reduces power losses at high temperatures, and a broader absorption range that includes ultraviolet and infrared radiation (Barragán Sánchez-Lanuza et al., 2024; C. B. et al., 2021). These characteristics give CIGS technology an advantage over crystalline silicon and other thin-film technologies. As the solar PV market grows, there is increasing attention on flexible PV technologies that allow for less fragile modules that can be rolled up and easily transported (Li et al., 2021), leading to lower installation time and costs (Ramanujam et al., 2020), and versatility in installation location and shape, like rooftops of houses and cars (Dallaev et al., 2023).

Despite these advantages of CIGS PV, challenges remain in terms of material availability and production costs (Maalouf et al., 2023). Indium, being an energy-critical element, raises concerns about the sustainability of metal reserves (Lee et al., 2024). Additionally, copper is toxic, and gallium is an expensive metal that must be managed wisely (Liu et al., 2022; Teknetzi et al., 2023). As the global market share of this thin-film solar technology continues to grow, the number of panels reaching the end of life (EoL) correspondingly rises (Komoto et al., 2022). To address

resource scarcity and reduce toxic waste from CIGS PV, it is crucial to determine efficient recycling methods for recovering critical elements. Evaluating the environmental and economic viability of recycling CIGS PV is also important for promoting circular economy models and ensuring the sustainable development of CIGS PV technology.

Recycling of CIGS PV involves the delamination of encapsulant(Deng et al., 2022; Dias et al., 2021; Meena et al., 2020) followed by material separation and purification techniques(Komoto et al., 2022; Komoto and Lee, 2018; Witto, 2023). Most previous recycling studies on CIGS PV have primarily utilized thermal or mechanical treatment options for delamination, combined with hydrometallurgical or pyrometallurgical methods and chemical leaching for the extraction of copper, indium, and gallium (Hu et al., 2022; Teknetzi et al., 2023; Witto, 2023). Leaching has emerged as the most common method, with sulfuric acid, hydrochloric acid, and nitric acid frequently used as leaching agents in nearly all experimental studies (Teknetzi et al., 2023; Witto, 2023). Additionally, recycling techniques for spent CIGS and chamber waste have been explored to recover valuable metals with higher extraction efficiencies (Hu et al., 2022; Li et al., 2022; Lv et al., 2019). However, very few studies have analyzed the environmental impact of these recycling approaches, and economic analyses regarding the recycling of CIGS PV panels are largely missing (Amato and Beolchini, 2019; Celik et al., 2020; Marchetti et al., 2018; Rocchetti and Beolchini, 2015).

In this study, we conducted a comparative assessment of the economic and environmental aspects of various EoL management techniques developed for CIGS PV. Through a systematic literature review, we identified and prioritized recycling procedures that are scalable, cost-effective, and

environmentally friendly (Amato and Beolchini, 2019; Liu et al., 2022; Marchetti et al., 2018; Rocchetti and Beolchini, 2015; Witto, 2023). Methods that use large amounts of leaching agents and release toxic gases have been eliminated (Gu et al., 2018; Gustafsson et al., 2015). Additionally, we excluded methods focused on spent CIGS recycling due to differences in composition and metal purity between spent CIGS and CIGS PV panels (Hu et al., 2022; Lv et al., 2019). We focused on recycling methods that provide data transparently for the entire procedure and are suitable for industrial scale recycling. These analyses highlighted three approaches with potential for mass production: the double-green process proposed by Marchetti et al. (Marchetti et al., 2018), the innovative method developed by Rocchetti and Beolchini (Rocchetti and Beolchini, 2015), and the high-yield recycling method developed by Liu et al. (Liu et al., 2022). For the selected recycling studies, we modeled recycling procedures, transportation, and waste disposal using the data collected from previous studies and based on our assumptions. Next, we calculated the economic and external costs for selected methods for the entire EoL of CIGS PV. Also, we identified the critical hotspots impacting the economic and environmental aspects of EoL of CIGS PV. Finally, after determining the economic benefit of the recovered materials, we calculated the net cost of recycling for three approaches by subtracting this benefit from the sum of private and external costs.

2. Methods

2.1 Goal and Scope

This study aims to compare the economic and environmental impacts of CIGS PV recycling processes. The results of this study will provide information on the combined economic and external costs for three CIGS PV recycling processes to increase knowledge of the feasibility of

potential PV recycling processes. The intended audience of this study is the recycling industry, the PV technology industry, consumers, and policymakers. The results of this study are intended to be used in comparative assertions and disclosed to the public.

In this life cycle assessment (LCA), we followed the ISO 14040/44 guidelines (ISO 14040:2006, 2006; ISO 14044:2006, 2006). There are two commonly used functional units in PV recycling LCA studies: 1) the amount of recycled materials (Kreiger et al., 2013; Latunussa et al., 2016), or 2) the surface area of waste PV panels (Lunardi et al., 2018; Rodriguez-Garcia et al., 2021; Tian et al., 2021). We selected the surface area (1 m² of CIGS PV module) in this study as a functional unit to provide a more straightforward interpretation of our life cycle inventories and impact assessment of our study. The system boundaries of this analysis include the EoL stage of the CIGS PV panels, and any transportation associated with the EoL. Recipe 1.08 (Goedkoop et al., 2013) and IPCC's GWP100 (Ecoinvent, 2024) impact assessment methods were used in this study. The environmental impact categories analyzed include climate change, ozone depletion, human toxicity, photochemical oxidant formation, particulate matter formation, ionizing radiation, acidification, freshwater eutrophication, marine eutrophication, terrestrial ecotoxicity, freshwater ecotoxicity, marine ecotoxicity, and land use. We also conducted the "normalization" optional elements of the life cycle impact assessment and converted the environmental impacts into external cost units. The database used to create three recycling processes LCA models was Ecoinvent 3.8 (Ecoinvent, 2023). openLCA V 2.0 is the software used to model all three recycling approaches (GreenDeLTa, 2023).

2.2 Description of selected recycling processes

We assessed three processes developed for CIGS PV recycling. Marchetti et al. rely primarily on mechanical separation, minimizing the number of chemicals used in the hydrometallurgical step:

leaching and centrifugal extraction. Rocchetti and Beolchini combine mechanical and hydrometallurgical steps, and the recovery of the valuable materials—indium and gallium—is achieved through electrodeposition. Liu et al. involve the physical separation, acid leaching, and individual metal separation and recovery of valuable metals.

Figure 1. a shows the flow diagram of Marchetti et al. (Marchetti et al., 2018). At the end of its life, the CIGS PV panel is taken by an automatic feed to the hammermill, where it is crushed into small pieces to facilitate recycling glass and break down the bonds of the encapsulant glue that holds it together. In the dimensional reduction stage, an industrial vacuum reduces dust pollution from the breakdown of the module. The larger pieces are then sent to the rotating drum, where the semiconductor film is leached using sulfuric acid (H_2SO_4) so that the metals end up suspended in the liquid. The newly leached module then goes to a centrifugal extractor to separate the liquid from the glass by sending the latter up the incline of a rotating screw and leaving the liquid to drain to the precipitation. The glass is sent to a vibrating screen, separating it from the encapsulant. Afterward, the glass is washed so any residual semiconductor film is completely removed and thus can then be shipped to a glass recycling facility. The liquid from the washing stage is transferred to a precipitation unit with sodium hydroxide (NaOH). Once the metals have been precipitated, the mud-like mixture of metals is transported to a filter press, where the excess water and waste are filtered out to leave the solids containing the metals.

In Figure 1. b, we show the steps of Rocchetti and Beolchini (Rocchetti and Beolchini, 2015), based on the Drinkard, Jr., Long, Goozner, and Ferron patents (Drinkard, Jr. et al., 1998; Ferron, 2012) to recycle CIGS PV panels. This process starts with a crushing phase in which a hammer

mill is used to reduce the size of the module. The crushing should reduce the module into 1-inch to 2-inch pieces. The pieces are then brought to a leach drum. In the leach drum, H_2SO_4 , hydrogen peroxide (H_2O_2), and an unidentified surfactant are added to oxidize gallium and indium. The glass and ethylene-vinyl acetate (EVA) are separated from the acidic liquid in a skimming and filtration step. Then, the glass was brought to a glass recycling facility, and the EVA was taken to a waste disposal facility and then to a landfill. Sulfur dioxide (SO_2) is used for the precipitation of selenium from the CIGS PV absorber layer. The next step is to adjust the pH, with NaOH, to be between 0.5 and 0.6 for indium to extract (Ferron, 2012). It must be filtered again to remove leach residue and solids. Next, di-(2-ethylhexyl) phosphoric acid (D2EPHA) and toluene are added to the filtered solution to make an indium organic acid that is then stripped with hydrochloric acid (HCl) to form an indium salt, so the metal extracted by the subsequent electrodeposition indium is pure. The solution goes through electrodeposition to remove the indium. Once the indium has been extracted, the solution is adjusted with NaOH to a pH range of 1.5-2.5 again, D2EPHA and toluene are added again to make an organic gallium acid, so it is separate from the leach solution. The solution is stripped with HCl to make gallium salt to increase gallium metal yield. The stripped gallium forms a solid and is extracted from the solution through electrodeposition.

Figure 1. c illustrates the steps involved in the recycling approach proposed by Liu et al. (Liu et al., 2022). In this recycling process, the CIGS PV panels that reached the end of their life were cut into 20cm square shapes and immersed in liquid nitrogen to separate the top tempered glass with molybdenum back contact by inducing thermal strain on EVA. The separated glass was sent to a glass recycling facility, and the EVA waste was subsequently sent to a waste disposal facility and then to a landfill. The CIGS PV absorber layer was separated from the EVA using HCl and

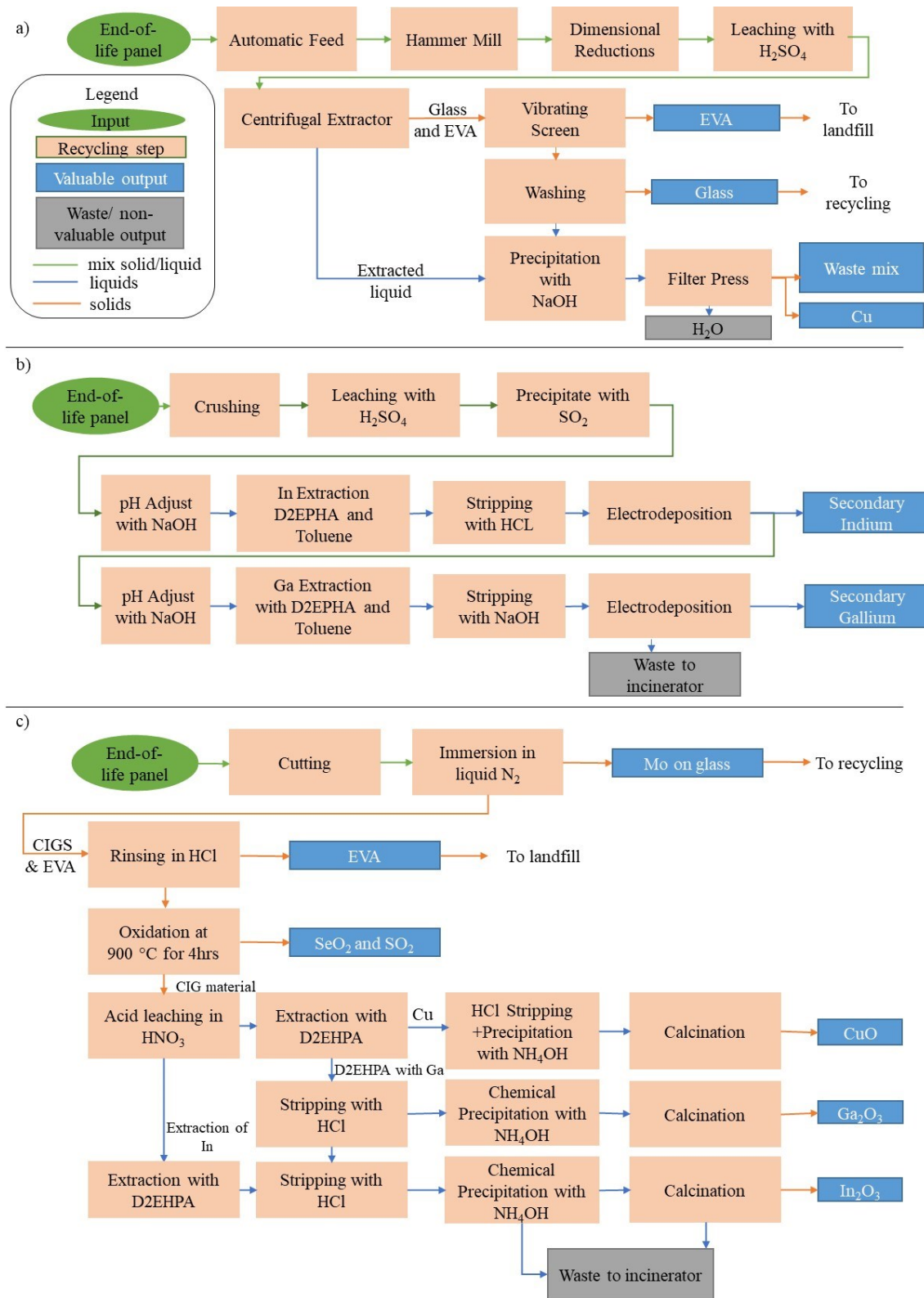
annealed in an oxygen atmosphere at 900 °C for 4 hours. During this oxidation process, metals formed metal oxides and selenium and sulfur were oxidized to gaseous forms. The oxides of copper, indium, and gallium were subjected to acid leaching with nitric acid (HNO₃) and solvent extraction with D2EHPA. The extraction with the D2EHPA process was conducted twice to extract the indium and gallium from the nitric solution efficiently. The organic forms of copper, indium, and gallium ions extracted were stripped with HCl and precipitated with ammonium hydroxide (NH₄OH) to form metal hydroxides. In the final stage, the metal hydroxides formed are sent for calcination and converted into metal oxides. The recovery rate of copper, indium, and gallium from these metal oxides is more than 90%.

2.3 Inventory analysis

We developed detailed life cycle inventories for EoL management of three CIGS PV recycling procedures, breaking down the data into process, transportation, and waste disposal steps, as shown in Table 1. The recycling process component includes equipment cost, electricity, and chemicals used for operations. The size of the CIGS PV modules is 1.4 m², and their weight is 24.18 kg (Marchetti et al., 2018). We, however, reported our results per m² to facilitate the comparison with other studies. We considered that both processes could treat 8,000 metric tons of waste per year (Latunussa et al., 2016), and the equipment would last 25 years (Markert et al., 2020). The equipment costs are reported per m² treated. Secondary data was collected from the literature regarding electricity and materials used in each recycling process (Liu et al., 2022; Marchetti et al., 2018; Rocchetti and Beolchini, 2015). The amount and kind of chemicals used for the panel treatment were taken from the literature and calculated their quantities per m² of the panel recycling. The electricity required for the operations in these recycling procedures was modeled for industry-scale manufacturing.

1
2 For transportation and waste disposal steps, we developed our inventories based on similar PV
3 recycling studies in the literature. Specifically for transportation, our modeling was based on
4 Latunussa et al. (2016): waste panels are transported 100 km from consumers to a collection point
5 and, from there, 400 km to the recycling plant (Latunussa et al., 2016). The waste generated during
6 recycling, such as used chemicals, is then transported 200 km to the waste treatment facility, and
7 the waste remaining after treatment is transported another 50 km to a landfill. The process waste
8 generated during the recycling processes of Marchetti et al. and Rocchetti and Beolchini was
9 calculated based on the information provided in these studies (Marchetti et al., 2018; Rocchetti and
10 Beolchini, 2015). For Liu et al. recycling process, we modeled waste generated during recycling
11 (Liu et al., 2022).

12
13 The associated cost inventories required for the economic assessment are provided in Table 2.
14 These inventories were modeled for the equipment and chemicals used in the recycling and they
15 were scaled up following a literature study (McCalmont et al., 2023). Transportation costs were
16 calculated using average freight revenue per ton-mile values provided by the United States
17 Department of Transportation (U.S. Department of Transportation, 2024). Waste disposal cost
18 includes waste treatment and disposal of the waste to landfill (Department of Toxic Substance
19 Control, 2024).



1

2 **Figure 1.** Flow diagram of a) Marchetti et al. (Marchetti et al., 2018) b) Rocchetti and Beolchini

3 (Rocchetti and Beolchini, 2015) and c) Liu et al. (Liu et al., 2022) recycling processes.

1 **2.4 External Cost**

2 In addition to the private cost of the CIGS PV panel recycling, we assessed the external cost. The
3 midpoint environmental impact emissions were calculated using ReCiPe 1.08 (Goedkoop et al.,
4 2013) and IPCC's GWP100 impact assessment methods (Ecoinvent, 2024). These emissions were
5 monetized using the external cost multipliers listed in Table 2. To calculate the external cost for
6 impact categories, we used midpoint environmental prices from (De Bruyn et al., 2018) and
7 updated by considering the inflation rate (Inflation Tool, 2024).

Table 1 Inventories for EoL management of CIGS PV including processing, transportation, and waste disposal for the selected recycling approaches. Process inventories are further classified into equipment, chemicals, and electricity used for the operations (denoted with E). T refers to transportation and W refers to waste disposal inventories.

			Units	Marchetti et al.	Rocchetti and Beolchini	Liu et al.
Process	Equipment	Conveyor belt	unit	1.73E-07	-	-
		Hammermill	unit	8.64E-08	8.64E-08	8.64E-08
		Vacuum	unit	8.64E-08	8.64E-08	-
		Vibrating screen	unit	1.73E-07	-	-
		Leach tank	unit	8.64E-08	8.64E-08	8.64E-08
		Classifier	unit	8.64E-08	-	-
		Screen washer	unit	8.64E-08	-	-
		Thickening tank	unit	8.64E-08	8.64E-08	8.64E-08
	Chemicals	Filter press	unit	8.64E-08	1.73E-07	8.64E-08
		Muffle furnace	unit	-	-	
		Electroplating machine	unit	-	1.73E-07	-
		H ₂ SO ₄	g	2571	750	-
		NaOH	g	2571	1335	-
		HCl	g	-	750	4619
		H ₂ O ₂	g	-	23.0	-
		Surfactant	g	-	23.0	-
		Toluene	g	-	265	-
		D2EPHA	g	-	66.0	70.0
		SO ₂	g	-	75.0	-
		Liquid N ₂	g	-	-	12.5
		HNO ₃	g	-	-	60.0
		NH ₄ OH	g	-	-	19.0
		CO ₂	g	-	-	25.0
	E	Electricity for operations	kWh	0.19	6.38	0.02
T	T	Transportation	tonne*km	9.45	9.50	9.80
W	W	Waste for disposal	kg	4.00	4.27	5.80

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Table 2 Cost inventories of EoL management of CIGS PV including processing, transportation, and waste disposal for 1 m² of module recycling. E denotes electricity, T denotes transportation and W denotes waste disposal cost inventories.

			Units	Cost	Reference
Process	Equipment	Conveyor belt	\$/unit	640	(Dongguan Xinshen Automation Technology Co.Ltd, 2024)
		Hammermill	\$/unit	4500	(Xi'an Rj Mechanical Equipment Co.Ltd, 2024)
		Vacuum	\$/unit	1899	(Suzhou Bersi Industrial Equipment Co.Ltd., 2024)
		Vibrating screen	\$/unit	1590	(Xinxiang Karp Machinery Equipment Co.Ltd., 2024)
		Leach drum	\$/unit	1600	(Qixia Dali Mining Machinery Co.Ltd., 2024)
		Classifier	\$/unit	5000	(Henan Baichy Machinery Equipment Co.Ltd., 2024)
		Screen washer	\$/unit	1000	(Hebei Jinghong Hydraulic Machinery Co.Ltd., 2024)
		Thickening tank	\$/unit	1000	(Wenzhou Qiangzhong Machinery Technology Co.Ltd., 2024)
		Filter press	\$/unit	6000	(Guangxi Jinbangtai Technology Co.Ltd., 2024)
		Electroplating machine	\$/unit	2000	(Junan Tongda Electronic Equipment Factory, 2024)
		Muffle furnace	\$/unit	6750	(Thermo Fisher Scientific Inc., 2024)
	Chemicals	H ₂ SO ₄	\$/kg	0.16	(ChemWorld.com, 2024)
		NaOH	\$/kg	0.41	(ChemAnalyst, 2024a)
		HCl	\$/kg	0.20	(ChemAnalyst, 2024b)
		H ₂ O ₂	\$/kg	0.75	(ChemAnalyst, 2024c)
		Surfactant	\$/kg	3.73	(ChemCentral, 2024)
		Toluene	\$/kg	1.05	(ChemAnalyst, 2024d)
		D2EPHA	\$/kg	1.10	(Jinan Future Chemical Co.Ltd., 2024)
		NH ₄ OH	\$/kg	1.32	(IndexBox Inc., 2024)
		HNO ₃	\$/kg	0.3	(ChemAnalyst, 2023)
		Liquid N ₂	\$/kg	1.18	(Rutherford & Titan, 2024)
		CO ₂	\$/kg	0.37	(IndexBox Inc., 2023)
		SO ₂	\$/kg	4.66	(Zibo Dijia Special Gas Co.Ltd., 2024)
	E	Electricity price (US)	¢/kWh	8.00	(U.S. Energy Information Administration, 2023)
W	W	Waste disposal fee	\$/ton	98.5	(Department of Toxic Substance Control, 2024)
T	T	Transportation	\$/t*mile	0.26	(U.S. Department of Transportation, 2024)

Table 2 External cost multipliers for environmental impact categories, according to (De Bruyn et al., 2018).

Impact Category	Unit	External cost Multiplier (\$/unit)
Climate change	Cost/kg CO ₂ -eq.	\$0.08
Ozone depletion	Cost/kg CFC-eq.	\$40.9
Human toxicity	Cost/kg 1,4 DCB-eq.	\$0.22
Photochemical oxidant formation	Cost/kg NO _x -eq.	\$20.2
Particulate matter formation	Cost/kg PM _{2.5} -eq.	\$53.8
Ionizing radiation	Cost/kg kBq Co ₆₀ -eq.	\$ 0.06
Acidification	Cost/kg SO ₂ -eq	\$ 6.72
Freshwater eutrophication	Cost/kg P-eq.	\$2.56
Marine eutrophication	Cost/kg N	\$4.20
Terrestrial ecotoxicity	Cost/kg 1,4 DCB-eq.	\$12.0
Freshwater ecotoxicity	Cost/kg 1,4 DCB-eq.	\$0.05
Marine ecotoxicity	Cost/kg 1,4 DCB-eq.	\$0.01
Land use	Cost/m ² *a crop-eq	\$0.13

2.5 Benefits of Recycling

Ideally, the costs of recycling CIGS PVs should be balanced by the benefits of recovering some of the valuable materials that CIGS PV waste contains. These include active materials such as indium, gallium, and copper. The input and output quantities of each recycling process are presented in Table 4. The input material quantities for the three recycling approaches were the same, calculated following Frischknecht et al. (Frischknecht et al., 2020). Marchetti et al. approach does not incorporate the extraction of indium and gallium separately after recycling, instead, they remain in the waste mix. Therefore, our analysis focuses only on the recovery of glass and copper for the Marchetti et al. approach. Rocchetti and Beolchini did not describe the treated modules or their recycling efficiencies except to indicate their process does not recover copper. So, we excluded copper when calculating the cost of recovered materials. We assumed the recycling efficiencies of

each process as 90% for calculating the output quantities of active materials and 95% recovery rate for glass at the end of recycling(Marchetti et al., 2018; Witto, 2023). For the Liu et al. recycling process, the output quantities were calculated based on extraction efficiencies provided in their study (Liu et al., 2022). To calculate the economic benefits of recovered materials, the cost inventories are provided in Table 4.

Table 4 Valuable inputs and outputs per m² of module, and costs of recovered materials. * Indicate the cost of glass cullet rather than the cost of glass used in PV synthesis. Additional information for extracting indium and gallium from the waste mix in Marchetti et al. and the benefits of these materials' recovery are also provided in SI.

Material	Marchetti et al.		Rocchetti and Beolchini		Liu et al.		Recovered Material Cost	Reference
	Inputs	Output	Inputs	Output	Inputs	Output	\$/unit	
Glass	17.0 kg	16.1 kg	17.0 kg	16.1 kg	17.0 kg	16.1 kg	10/ton*	(ScrapMonster, 2024)
Indium	3.43 g	-	3.43 g	3.09 g	3.43 g	3.28 g	400/kg	(USGS National Minerals Information Center, 2024)
Gallium	1.72 g	-	1.72 g	1.54 g	1.72 g	1.64 g	618/kg	(USGS National Minerals Information Center, 2024)
Copper	12.0 g	10.8 g	-	-	12.0 g	11.1 g	8.32/kg	(USGS National Minerals Information Center, 2024)

3. Results

3.1 Private cost of recycling

Figure 2 compares the private cost breakdown for three recycling processes. Our results show that the recycling process developed by Rocchetti and Beolchini is the most expensive option, while Marchetti et al. have the lowest private cost. In the Rocchetti and Beolchini process, the process cost emerges as the largest contributor among the components of CIGS PV's EoL management, while transportation costs were identified as the predominant factor in Marchetti et al. and Liu et

al.'s recycling approaches. Waste disposal costs are insignificant for all three recycling approaches.

The total private cost of EoL management of three recycling processes varies from \$3.5 to \$4.5 per m². The private cost of the Rocchetti and Beolchini is \$4.24/m², ~ 15% greater than Marchetti et al.'s (\$3.60/m²) and Liu et al.'s (\$3.70/m²) methods. For both Rocchetti and Beolchini and Marchetti et al. approaches, the process cost dominates the total private cost, accounting for nearly 50%. The chemicals employed in recycling contribute to the higher process cost in these two approaches. For Rocchetti and Beolchini, H₂SO₄, HCl, NaOH, SO₂, and a small amount of other chemicals make up ~40% of the total private cost. The electricity utilized in Rocchetti and Beolchini is the second most significant contributor to the process cost, accounting for ~12% of the total private cost, with 94% of the electricity costs attributed to the high energy required by electroplating. In the case of the Marchetti et al. approach, NaOH and H₂SO₄ employed in the recycling process constitute the majority (~50%) of the private cost. Regarding Liu et al.'s private cost, transportation cost (\$1.8/ m² of PV panel) is dominant, constituting ~40% of the private cost. The increased transportation cost is associated with waste disposal generated throughout the recycling process (5.8 kg/m²), nearly 1.5 times higher than the waste generated in the Marchetti et al. and Rocchetti and Beolchini approaches. This is attributed to the waste produced from excessive utilization of HCl in the initial separation of EVA from the CIGS absorber layer.

We compared our private recycling cost with the recycling cost of other PV technologies, e.g., monocrystalline silicon (c-Si) PV and thin film perovskite PV, as there is a lack of studies on the cost of CIGS PV recycling. According to the US National Renewable Energy Laboratory, the cost

of PV module recycling in the United States is \$15- 45 per module(Curtis et al., 2021). Our modeling results for CIGS PV modules recycling costs are less than this range across three recycling approaches.

Markert et al. investigated the recycling of c-Si PV panels using the full recovery of the EoL photovoltaic project method (Markert et al., 2020). The study evaluated the private cost of recycling across process, transportation, and waste disposal categories, similar to our approach. Markert et al. found that to recycle 1 m² of c-Si PV panel, the total EoL private cost would be \$6.72/m² (Markert et al., 2020). Our modeling results for three recycling approaches of CIGS PV show ~ 1.5 to 2 times lower private cost than this value. Markert et al. found that the largest contributor to the private cost was transportation (50%), and the smallest was the recycling process (4%). In contrast, our results reveal that the two processes followed a similar pattern, with process cost (~50%) being the main contributor, followed by transportation (~40%) and waste disposal (~10%). This difference can be attributed to the increased usage of chemicals in our modeled recycling procedures, and the transportation distance considered in our modeling is 750 km compared to 850 km in their study. Similar to Markert et al., our results for Liu et al. indicate that transportation (~50%) contributes more significantly to overall private costs than process costs. This is attributed to the higher amount of waste generated during the recycling process, necessitating increased transportation costs, while the quantity of chemicals is comparatively lower. Our waste disposal cost in all three recycling approaches is significantly lower (\$0.40- \$0.66) than that of Markert et al. (\$3.11). This difference arises from Markert et al. considering different tipping fees for various chemicals, such as contaminated glass, fly ash, liquid waste, and sludge, compared to the single disposal cost for the waste disposal (\$98.5/ton) in our study.

Furthermore, we compared our findings with the PSC module recycling cost as Chen et al. modeled. (Chen et al., n.d.). According to their results, the recycling process cost of PSC modules is \$4.24/m², which is comparable to our process cost for three recycling approaches modeled for CIGS PV.

3.2 External cost of recycling

The total external cost of the Rocchetti and Beolchini approach is \$3.90/m², ~1.3 times higher than Marchetti et al. (\$3.04/m²) and Liu et al.'s (\$3.16/m²). The external cost breakdown by impact source of all three recycling processes is shown in Figure 3. Recycling process external costs are contributing most (~70 to 80%), followed by waste disposal (~10 to 20%) and transportation (~10 %) in all three recycling approaches. The external cost of Marchetti et al. comes from environmental impacts associated with using chemicals NaOH and H₂SO₄ in the recycling process. For the Rocchetti and Beolchini approach, chemicals used in the recycling process, mainly NaOH, H₂SO₄, HCl, and electricity used for multiple electrodeposition steps, contribute to this higher external cost. Liu et al. external cost is dominated by HCl used in the recycling process and waste disposal.

We further compared our data with the literature. The external cost of c-Si's was reported to be \$5.7/m². Our external cost for three CIGS PV recycling approaches is lower than this value. This difference is attributed to using different chemicals and energy inputs in recycling. Similar to our results, Markert et al. found that the largest contributing source to the external cost was the chemicals employed in the process, which made up 72% (Markert et al., 2020). This was followed by transportation at 13% and disposal at 6%. Note that the transportation distances in the EoL

- 1 phase reported by Markert et al. were longer than our study's, making the transportation-associated
- 2 external cost higher than the waste disposal cost.

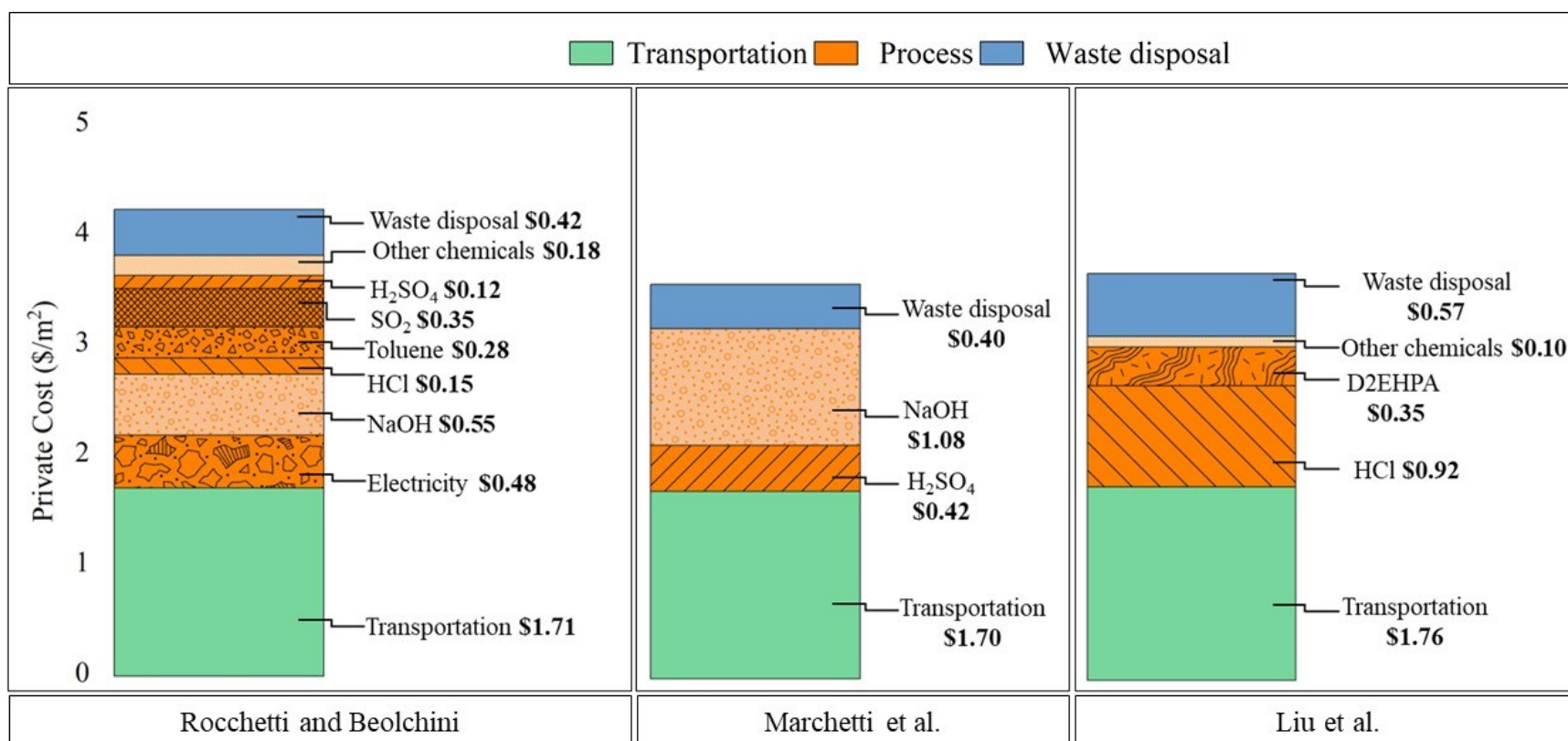
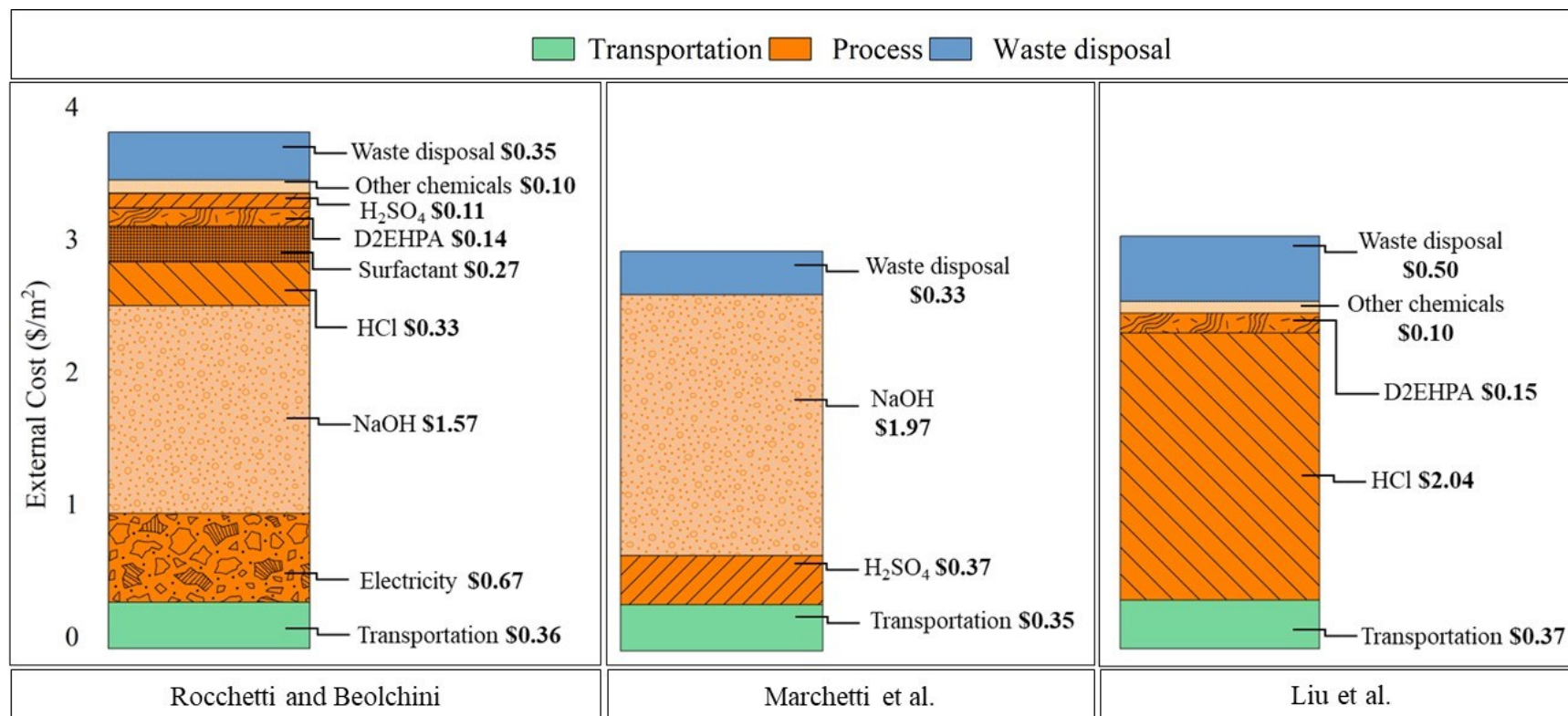


Figure 2 Private cost breakdown of CIGS PV EoL management for three recycling processes.

1



2

3 **Figure 1** External cost divided by impact source for three recycling approaches.

Figure 4 shows the breakdown of external costs divided by impact category for three recycling approaches. The three recycling approaches exhibit variations in the most significant impact category influencing their external cost, with climate change, particulate matter (PM_{2.5}) formation, and photochemical oxidants identified as commonly contributing significantly to the overall external cost. For the Marchetti et al. approach, PM_{2.5}, photochemical oxidants, and climate change emissions make up ~70% of the total external cost. About 50% of climate change, 60% of photochemical oxidants, and more than 90% of PM_{2.5} emissions are associated with NaOH used in precipitation metals in the Marchetti et al. approach. Transportation is the second most significant contributor (~20%) to these three major impact categories, with upstream emissions from diesel fuel extraction in the refinery accountable for these emissions across all three impact categories. For the Rocchetti and Beolchini process, freshwater eutrophication stands out as a dominant external cost factor, comprising 25% of the total, followed by climate change, photochemical oxidant, and marine eutrophication, each contributing nearly 15% to the overall external cost. The excessive use of NaOH for pH adjustment and stripping stages is accountable for higher environmental emissions (~25 to 45%) in these categories. Both freshwater and marine eutrophication external costs are also dominated by electricity (~30 to 40%) used in multiple electrodeposition steps. In addition to NaOH, climate change impacts are also dominated by waste incineration (30%) and transportation (21%). For Liu et al.'s recycling process, emissions contributing to climate change constitute ~30% of the overall external cost, followed by the photochemical oxidant and freshwater eutrophication impact categories, each accounting for ~20%. Notably, the utilization of HCl in acid leaching and stripping stages significantly influences the external cost, accounting for ~50 to 90% among these impact categories.

The assessment of the external cost of PV recycling is limited. Markert et al. assessed c-Si PV recycling and found that the largest contributing environmental impact category was acidification (46%), followed by ecotoxicity (28%) and PM (9%). This differed from our study, which found climate change, PM_{2.5}, photochemical oxidant with the largest impact, and acidification with a small impact. The high acidification costs for the full recovery of the EoL photovoltaic project process are associated with sieving and acid leaching, as well as filtration, electrolysis, and neutralization(Markert et al., 2020). Conversely, the environmental impacts of the three recycling approaches are associated with chemicals used in precipitation, stripping, metal extraction, and waste disposal stages.

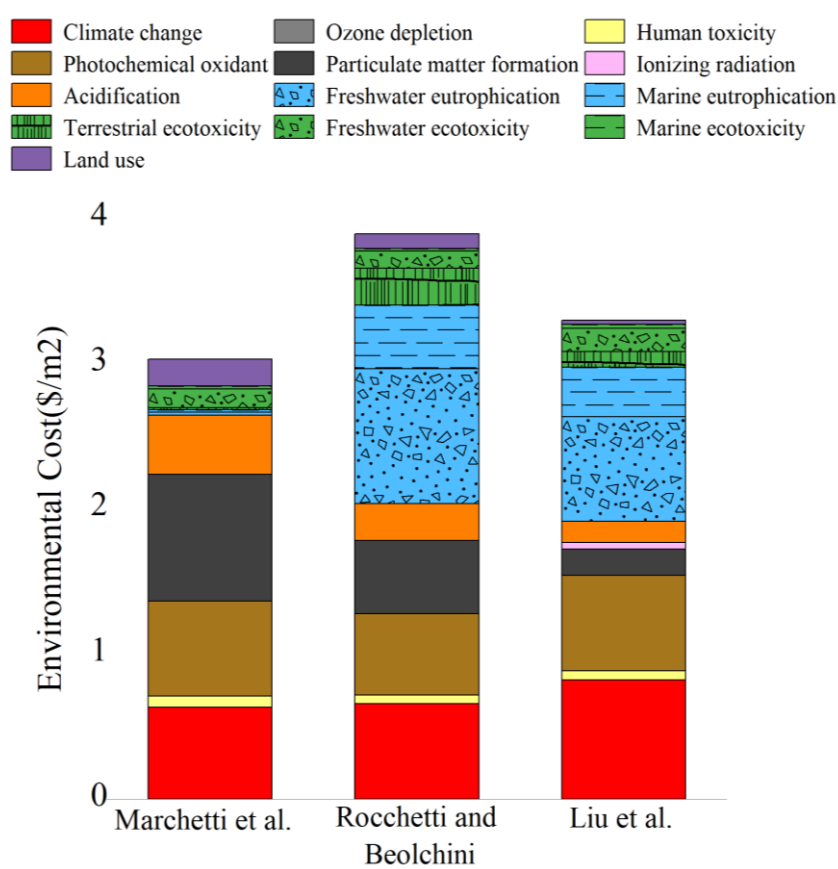


Figure 2 External cost breakdown by impact category for three recycling approaches.

3.3 Net cost of CIGS PV Recycling

The breakdown of CIGS PV recycling across three recycling approaches is shown in Figure 5. The net cost analysis harmonizes private and external costs and revenue generated (Markert et al., 2020); positive values indicate the private and external costs, while negative values indicate the revenue generated through three material recovery approaches. The economic value derived from recovered materials is \$ 1.3, \$ 3.4, and \$ 3.6 per m² in the CIGS PV recycling proposed by Marchetti et al., Rocchetti and Beolchini, and Liu et al., processes, respectively. The breakdown of revenue generated from the materials recovery of Marchetti et al. is 93% (glass cullet) and 7% (copper). In comparison, in the Rocchetti and Beolchini process, 36% (indium), 35% (glass cullet), and 28% (gallium) contributed to revenue from recovered materials. Liu et al.'s process for materials recovery aligns with the breakdown obtained by Rocchetti and Beolchini, including an additional 2% revenue attributed to copper recovery.

The net recycling costs of the three recycling approaches are further compared with those of other PV technologies. Markert et al. found the net cost of c-Si recycling is \$-1.19/m², which resulted from deducting the economic benefit of recovered materials \$13.6/m² from the total cost of c-Si recycling, including both private and external costs is \$12.43/m² (Markert et al., 2020). This negative value of the net recycling cost indicates an economic benefit of \$1.19/m², even after considering external costs. Our net recycling costs for three recycling approaches are \$5.3/m², \$4.7/m², and \$3.2/m² for Marchetti et al., Rocchetti and Beolchini, and Liu et al., processes, respectively. In comparison to Markert et al., the net cost calculated for three CIGS PV recycling approaches in this study demonstrated limited economic benefits. This disparity is attributed to variations in the material recovered and efficiency differences between the two studies. For

example, the recovered materials value in Markert et al. primarily resulted from the recovery of aluminum at \$6.3/m²(Markert et al., 2020), while recycling of aluminum used in the CIGS PV module was not considered in three of the recycling processes examined in this study, as they exclusively focused on frameless PV modules. These findings indicate that efforts to recover CIGS PV panels should extend beyond active layer materials, prioritizing enhancement in material recovery to effectively compensate for private and external recycling costs. In a separate study by McCalmont et al., the PSC module's net recycling cost was \$-2.95/m² (McCalmont et al., 2023). The economic benefits of recovered materials in their study amount to \$13.65/m², 3 to 10 times higher than those observed in the three CIGS PV recycling approaches assessed in our study(McCalmont et al., 2023). The higher economic benefit in their study mainly stems from the recovery of coated glass recovery as unbroken (\$6.9/m²). In three studied recycling approaches, glass is mechanically crushed and recovered as glass culets. The economic benefit of the glass cullet recovered from three CIGS PV recycling processes is lower than that of solar glass recovered from McCalmont et al.'s study. An important additional point to note is that in the study conducted by McCalmont et al., the net cost value did not account for external costs.

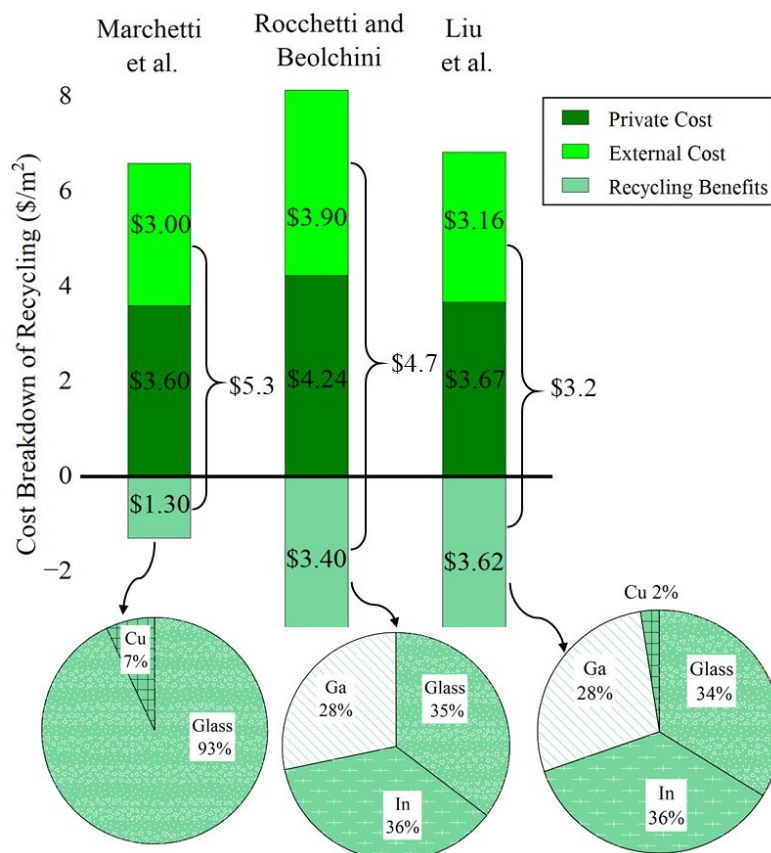


Figure 5 Total cost breakdown of three CIGS PV recycling approaches. Pie charts illustrate the percentage of benefits derived from each parameter in every recycling approach.

4. Conclusions

In this study, we assessed both the private and external costs of three EoL components—process, transportation, and waste disposal—for three CIGS PV recycling approaches. Our results show that the private cost of EoL management of CIGS PV ranges from \$3.5 to \$4.5 per m², while the external cost varies from \$3.0 to \$4.0 per m². The primary contributor to both the private and external costs was found to be the consumption of chemicals during the recycling process. NaOH and HCl usage in recycling significantly impacts climate change, photochemical oxidant

1 formation, particulate matter formation, and freshwater eutrophication contributing ~50% to 90%
2 of these impacts. Finally, our net cost analysis reveals limited economic benefits derived from the
3 CIGS PV recycling. This suggests that the efforts to recover materials should expand beyond active
4 layer materials, emphasizing enhancing materials recovery to adequately offset both the private
5 and external costs associated with recycling.

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12 **Author Contributions**

13 Conceptualization, I.C.; Methodology, A.R, E.G., and A.T; Data curation, A.R., E.G., and A.T.;
14 Writing—original draft, A.R, E.G., and A.T; Writing—review & editing, A.R., and I.C.;
15 Visualization, A.R., and E.G; All authors have read and agreed to the published version of the
16 manuscript.

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