REGIONAL PHOTOVOLTAICS CIRCULARITY UNDER VARIOUS SOCIO-ECONOMIC CONDITIONS

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Abstract

The United States (U.S.) Infrastructure Investment and Jobs Act of 2021 (IIJA) and the Inflation Reduction Act of 2022 (IRA) have been found to drive significantly increasing deployment of wind turbines and solar photovoltaics (PV) capacity (to 44 GW/year), leading to clean electricity shares of 70% by 2030. Such deployment of solar and wind assets poses end-of-life (EOL) management questions. To reach decarbonization goals, around 1.75 TW of cumulative PV capacity needs to be deployed in the US by 2050, which will generate about 8 million metric tons (Mt) of EOL PV materials. While the quantity and toxicity of PV waste is not as concerning as other type of waste, recovery of valuable and energy-intensive materials from EOL PV is expected to bring economic and environmental benefits. Indeed, building a circular PV industry would maintain high-value and critical PV materials such as aluminum, silicon, copper and others within the U.S. economy, enhancing its resilience and sustainability. However, the economics are not favorable: PV recycling costs about one order of magnitude higher than landfill costs based on best current understanding. More favorable economics highly depend on reverse supply chain infrastructure and regulations, which are not yet optimized for supporting PV circularity.

In this work, we present an analysis of EOL PV recycling rate under various transportation, recycling, and landfill cost conditions and regulatory contexts. Combining the National Renewable Energy Laboratory PV in the Circular Economy (PV ICE) and circular economy agent-based model for PV (CE ABM) models, we increased the accuracy of PV reverse supply chain modeling compared to our previous model, better capturing PV materials by vintage, transportation distances, and geographic resolution. Results show that transportation distances and associated costs strongly affect decisions to recycle or not. With more stringent regulations, recycling could be favored since both landfill and transportation costs could significantly increase. However, this is assuming that less stringent transportation and storage regulations applies for recycling. Moreover, results show that the mass-based recycling rate could be up to 75% under the hazardous waste regulation scenario. About 260,000 metric tonnes of Silicon is potentially recoverable during recycling between now and 2050 in this scenario, nearly enough to supply all the domestic Silicon demand from silumin (aluminum-silicon alloy) manufacturing for that same period.

Introduction and Motivation

There is a two-in-three chance of one of the next five years to cross the 1.5°C Global warming threshold (WMO 2023). In this context, actions are needed to reduce anthropic greenhouse gas (GHG) emissions. Recent laws enacted by the United States (U.S.) Congress – the Infrastructure Investment and Jobs Act of 2021 (IIJA) and the Inflation Reduction Act of 2022 (IRA) – have the potential to significantly change the US power sector. A report by the National Renewable Energy Laboratory (NREL) has estimated that the IRA and IIJA could accelerate the deployment of renewable energy technologies and, by 2030, decrease the electricity grid GHG emissions by 80% compared to 2005 (Steinberg et al. 2023).

Although barriers (e.g., siting and permitting challenges, higher projected costs of renewable energy technologies, for instance, due to supply restrictions) could hinder renewable energy deployment, even taking these factors into account, the IRA and IIJA were found to drive the deployment of wind turbines and solar photovoltaics (PV) capacity (44 GW/year) – leading to a share of clean electricity of 70% by 2030 (Steinberg et al. 2023). However, deployment of solar and wind assets at this scale raises questions, including with regards to end-of-life (EOL) management. To reach the decarbonization goals that the IRA and IJJA support, around 1.75 TW of cumulative PV capacity needs to be deployed in the U.S. by 2050 (Mirletz et al. 2022). This level of deployment is estimated to generate about 8 million metric tons (Mt) of EOL PV materials from modules alone (Mirletz et al. 2022). While PV waste might not be as concerning as other type of waste (Mirletz et al. 2023), the recovery of valuable and energy-intensive from EOL PV – some of them being on the DOE (2023) critical materials list – is expected to bring economic and environmental benefits (Mirletz et al. 2023; Heath et al. 2020).

Review of Related Work

Crystalline silicon PV modules – which represents 97% of the global PV market – are made out of 10% aluminum and 3% silicon by mass (Mirletz et al. 2023), two materials that are considered near critical in the medium term (DOE 2023). Cadmium telluride (CdTe) PV modules – which represents 3% of the global PV market – are made out of 11% to 14% aluminum by mass (Mirletz et al. 2023). The demand for these materials is likely to grow (Sovacool et al. 2020) and so are the potential supply risks (DOE 2023). It would be helpful to improve the resilience of these material supply chains to mitigate risking the transition to a decarbonized US economy and climate change mitigation. Building a circular PV industry would maintain high-value and critical PV materials within the US economy, enhancing its resilience and sustainability (Walzberg, Carpenter, and Heath 2021).

The circular economy (CE) is often viewed as an economic model capable of mitigating the environmental and social impacts linked to natural resource exploitation. Circularity goes beyond material-level reprocessing (i.e., recycling), striving to keep materials and products at their highest economic value. The CE does so through four main pathways: by narrowing (using less natural resources), slowing (extending product lifetimes), regenerating (using renewable or regenerative resources), and closing (recycling materials and recovering energy) natural resource flows (Walzberg et al. 2025). CE measures could contribute to climate change mitigation and adaptation by reducing global greenhouse gas (GHG) emissions from material use and regenerate the natural environment (Wang et al. 2022). The CE could also bring about other environmental benefits, such as reduced land and water use, and toxicity (Walzberg et al. 2025).

While a circular PV industry would be beneficial, the economics are not favorable (Table 1). First, best current estimates (though incomplete and uncertain) indicate that PV recycling costs about one order of magnitude higher than landfill costs (Iakovou et al. 2024). Second, transportation costs for bulky and heavy items like PV modules (which measure roughly 2x3 m^2) can also be prohibitive, especially over long distances. Both landfill and transportation costs depend on how PV waste is regulated in addition to weight and volume (Curtis, Buchanan, Smith, et al. 2021). Besides economics, other factors affecting PV recycling are both technical (e.g., the material recovery efficiency of the recycling process and the purity of the materials that can be recovered through it) and non-technical (e.g., the complexity of navigating the regulatory framework or the perceived risk of choosing one option over another). Thus, PV circularity highly depends on the reverse supply chain infrastructure and its context, which have not been optimized yet.

Table 1: Costs of different end-of-life management option – adapted from Iakovou et al. (2024) and Wang et al. (2021).

End-of-life option	**Cost ($/module)**
Reuse	20-60
Recycling	10-45
Non-hazardous landfill	0.5-1.8
Hazardous landfill	3.6-5

The objective of this study is to present an analysis of EOL PV recycling under various transportation, recycling, and landfill cost conditions and regulatory contexts. The NREL PV in the Circular Economy (PV ICE) tool and circular economy agent-based model for PV (CE ABM) model are combined to provide an enhanced modeling of the PV reverse supply chain. The model captures PV material composition by vintage and maintains a high geographic resolution, enabling the calculation of more accurate transportation distances. The next sections present the methodology used for and the results obtained from the analysis.

Technical Approach

Circular Economy Agent-Based Model

The CE ABM framework developed during previous work is used for this study (Walzberg et al. 2023). In short, the framework aims to evaluate techno-economic, market, and social conditions maximizing circularity for different technologies. Several types of agents are defined depending on the technology, with asset owners, service providers, recyclers, and manufacturers being common to all instances of the CE ABM. In addition to techno-economic factors, the CE ABM framework accounts for behavioral aspects such as social norms and trust.

For PV, the CE ABM considers PV owner, installers, recyclers, and manufacturers, and five EOL options are modeled: repair, reuse, recycling, shredding, and storage (Walzberg, Carpenter, and Heath 2021). The primary output of the model is the mass of PV modules reaching each EOL option during the simulation period (2020- 2050 in this study). During the CE ABM simulations, agents interact in different ways. First, agents of the same type (e.g., PV owners) influence each other, thereby modeling the effect of peer influence and social norms on EOL management decisions. Second, information flows between agents of different types. For instance, installer agents have access to the stock of EOL PV modules handed over by PV owner agents.

The theory of planned behavior (TPB) is then used to model PV owners' purchase and EOL management decisions (Ajzen 1991). The TPB accounts for three main factors affecting the intention to perform a behavior: the perceived behavioral control (which measures the difficulty of performing the behavior), the attitude hold toward the behavior (which measures whether the behavior is perceived as favorable or unfavorable), and the subjective norms (which refers to the perceived social pressure to perform or not perform the behavior). Equation 1 summarizes how each individual EOL option is scored based on the TPB.

$$BI^{t}_{ij} = w_{A}A^{t}_{ij} + w_{SN}SN^{t}_{ij} + w_{PBC}PBC^{t}_{ij} \quad (1)$$

Where at any time step t, for each agent i and EOL option j, BI is the behavioral intention of performing the behavior, A is the attitude toward the behavior, SN represents the subjective norms, PBC is the perceive behavioral control over the behavior, and w_{A}, w_{SN}, and w_{PBC}, are weighting factors. Agents choose the EOL option which obtains the highest score according to equation 1.

To simplify the analysis, this study focuses on PV EOL management decisions and, therefore, PV purchase decisions are not included. Moreover, given the uncertainty of the various factors that could affect EOL PV management decisions, most of the results presented in the discussion were obtained with a model that only accounts for PBC. Those results also assume that PBC only relates to the financial costs of performing a behavior. In other words, for most of the results presented below, the agents' EOL management decisions strictly aim at minimizing costs. This modeling choice enables a more focused analysis and prepare work for future reverse supply chain optimization work we plan to perform. However, our previous study offered a deeper analysis of social factors affecting PV recycling and reuse (Walzberg, Carpenter, and Heath 2021).

Combining the Circular Economy Agent-Based Model with the PV in the Circular Economy tool

The PV ICE tool is a dynamic mass-flow analysis framework. The purpose of the tool is to enable mass-flow analyses, incorporating granular, evolving material baselines and estimate circularity in different pre-determined scenarios (Ovaitt et al. 2022). The system boundaries of PV ICE start at the extraction of raw materials and ends at end-of-life management and includes manufacturing and use of PV panels. The tool tracks the virgin material demands, manufacturing scrap, installed capacity (nameplate and effective), and EOL PV generated in a given scenario, both annually and cumulatively. Instead of using a fraction of module mass to derive material flows, PV ICE uses the annual average module efficiency and annual capacity deployments to calculate the module area installed each year.

From there, component material masses are calculated on a per-module area basis. Thus, the ''average module'' for each year is defined as a module with the weighted average energy yield and material composition of modules deployed in that year (defining the "vintage" for that year). This vintage is unique for each year due to PV technology evolutions, and it enables PV ICE to capture design changes in newer modules, efficiency and reliability improvements, and known material risks from specific years. More details on the model can be found in Ovaitt et al. (2022) and Mirletz et al. (2022).

To combine the CE ABM with PV ICE, several functions were added to the former source code (Figure 1). Since PV ICE presents itself as a Python package, it is easy to launch its functions from another Python model. First, PV ICE is used to read a NREL Regional Energy Deployment System (ReEDS) file, which contains annual capacity deployments. This input file can directly be changed (to different projections) within the CE ABM and from there, different PV ICE scenarios can be launched. Next, PV ICE acquires geographic information system (GIS) data for each power control areas (PCAs) and establishes a comprehensive list of outputs by scenarios, and PCAs from the original ReEDS file based on a dynamic material flow approach. To do so, the mass flow calculation function from PV ICE is used for each material, scenario, and PCA. These calculations yield two datasets that are subsequently used by the CE ABM: "data out," representing cumulative amounts encompassing all materials, and "matdataout," providing material-specific values up to the year 2050.

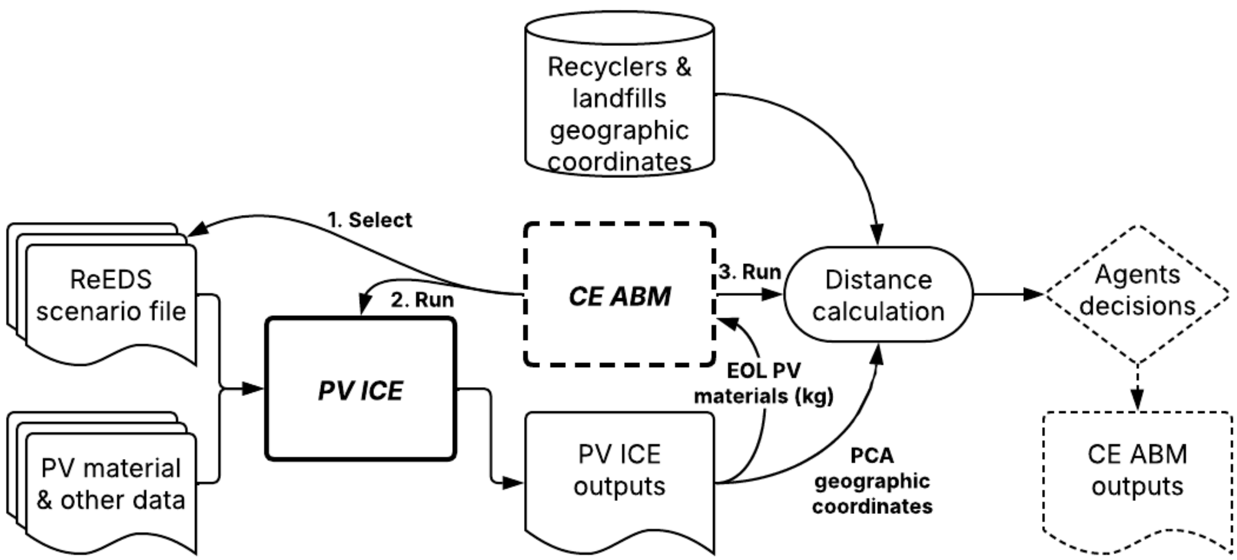

Figure 1: Combination of the PV ICE tool and CE ABM. Solid lines represent new additions (or major modifications) to the original CE ABM. Dashed lines represent the original CE ABM and functions that were not altered during the combination. Numbers represent the sequential steps leading to a successful run with the PV ICE-CE ABM integrated model.

The CE ABM code was modified to use the datasets generated by PV ICE instead of its own internal calculation (which as discussed previously are not as accurate and aggregated at the national level) – namely, the geographically- and temporally-explicit installed PV capacity and EOL volume data. The transportation cost function in the CE ABM was also improved to take advantage of the higher spatial resolution offered by PV ICE. To do so, PV recycler data from the Electric Power Research Institute (EPRI 2023) and landfill data from the Waste Business Journal (2020) and the Environmental Protection Agency EPA (2023) were obtained. When those datasets did not contain geographic coordinates but only city or zip code information, the Geopy Python library was used to obtain the missing longitudes and latitudes (Contributors 2014). Next, centroid coordinates for all PCAs were obtained from their respective GIS file (Ho et al. 2021) and the Haversine formula (Maria et al. 2020) was used to calculate the transportation distances between any PCA centroid (assumed to be the location of PV waste generation) and any landfill or recycling facility. Practically, the straight-line distance d between two sets of geographic coordinates (lat_1, $long_1$) and (lat_2, $long_2$) is calculated as follows:

$$a = \sin^2((lat_2 - lat_1)/2) + \cos(lat_1) \times \cos(lat_2) \times \sin^2((long_2 - long_1)/2) \quad (2)$$

$$c = 2 \times atan2(\sqrt{a}, \sqrt{(1-a)}) \quad (3)$$

$$d = R \times c \quad (4)$$

With *R* the radius of the earth. Applying equations 2-4 significantly improves the accuracy of the CE ABM transportation cost calculations, which was previously using state centroids and a shortest path algorithm (Walzberg, Carpenter, and Heath 2021).

Following those modifications, the CE ABM was used to estimate PV recycling rates (mass-based) under various socio-economic conditions – focusing especially on transportation, recycling, and landfill cost conditions and regulatory contexts. The next section presents and discusses results.

Discussion

The effect of transportation costs and the regulatory context

Figure 2 shows how different assumptions on PV waste regulatory status (either considered as hazardous or industrial solid waste) affect the 2050 PV cumulative recycling rate in the contiguous U.S. In this figure, it is assumed that recycling and landfill fees are equal so that only the effect of various transportation distances (and thus costs) are analyzed. It is worth noting the investigation presented in Figure 2 is based on hypothetical and extreme cases. As of now, it is the responsibility of the waste generator to determine if a module is hazardous or not through the Toxicity Characteristic Leaching Procedure (TCLP). We understand that, anecdotally, some modules have been classified as hazardous by the TCLP, and others have not. The objective of Figure 2 is to gain insights from those hypothetical cases.

The figure shows that. since there are only a few landfills accepting hazardous waste, but many more accepting industrial waste, EOL PV would need to be transported over longer distances if they were considered hazardous waste. Moreover, landfill fees would be more expensive. This would lead many PV owner agents to choose recycling as it would be the most economical option (Figure 2-A). In this scenario, the overall recycling rate is 75%.

On the contrary, as it can be seen on Figure 2-B, if EOL PV is considered industrial waste, transportation costs to landfill are, on average, much lower than for recycling. In that scenario, the recycling rate is 1.5%. The contrast between the two scenarios shows that – even without considering the higher costs of recycling compared to landfilling – the sheer number and proximity of landfill sites compared to recycling sites is enough to discourage recycling. It also shows that transportation distances and costs could be important factors to consider to encourage PV recycling.

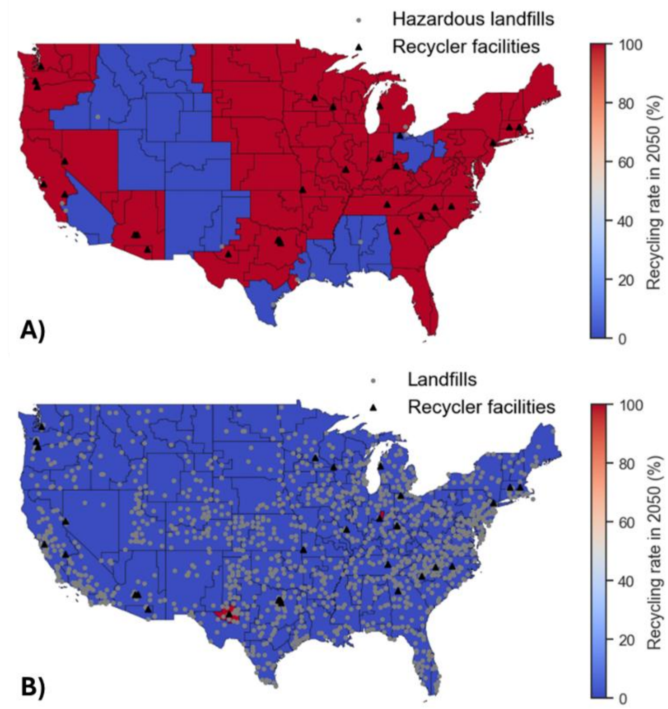

Figure 2: Cumulative recycling rate in 2050 across the U.S. electricity grid power control areas. The results shown in this figure assume recycling fees are equal to landfill fees so that only the effect of various transportation distances and costs are assessed. A) EOL PV are considered hazardous waste and can only be disposed of in hazardous landfill (grey dots on the map). Hazardous landfill costs are assumed $175/tonne. B) EOL PV are not considered hazardous waste and can be disposed of in any landfill (red dots on the map). Landfill costs are assumed $87/tonne. Transportation costs are assumed to be $1.5/t*km for both A) and B).

Given the lack of practical experience with EOL PV management (since most of the PV panels deployed in the U.S. have yet to reach their EOL), transportation costs are quite uncertain. Anecdotal evidence suggest that they sometime could represent as much as the cost of recycling the PV panels themselves (PV Circularity Workshop 2024). Thus, Figure 3 presents additional results when varying transportation costs. Different assumptions were used than in Figure 2. First, EOL modules are all considered industrial waste. Second, recycling costs are assumed to be $0/module to represent the most optimistic case for PV recycling (e.g., if the value of recovered materials were to equal the cost of recycling, or otherwise subsidized). Landfill locations are the same than in Figure 2-B, however, individual landfill fees from the Waste Business Journal (2020) were used instead of the average landfill fee calculated from that database ($87/tonne). Finally transportation costs were varied from $1.5/t*km (which translates to a maximum cost of about $30/module which is similar to reported recycling costs (Iakovou et al. 2024)) to $0/t*km.

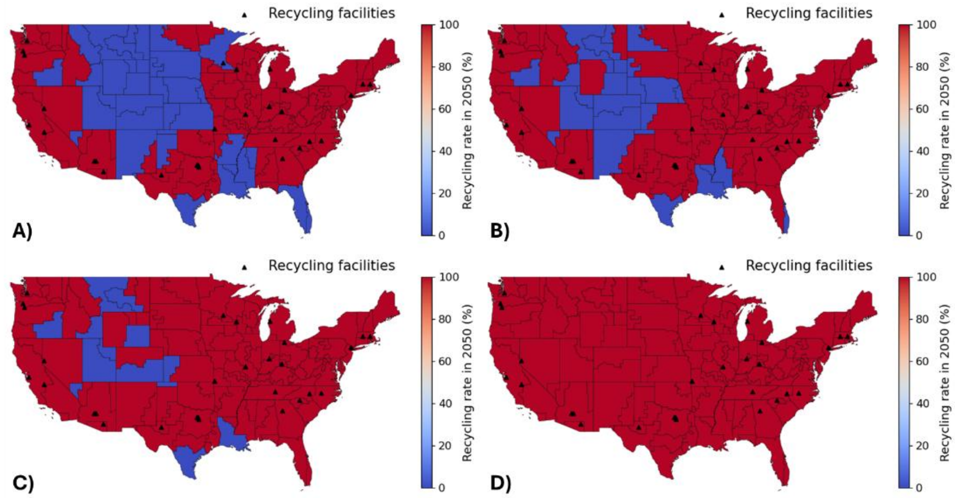

Figure 3: Cumulative recycling rate in 2050 across the U.S. electricity power control areas. In this figure, recycling fees are $0/module and reported individual landfill fees are used. A) transportation costs are $1.5/t*km. B) transportation costs are $1.0/t*km. C) transportation costs are $0.5/t*km. D) transportation costs are $0.0/t*km.

Figure 3-A shows that high transportation costs would be detrimental to PV recycling in most of the PCAs that do not have a recycling facility. As recycling costs decreases, recycling picks up. Interestingly, even when transportation costs are reduced to only $0.5/t*km, some landfills which have low fees can still be attractive, even in PCAs with recycling facilities nearby (see, for instance, the blue PCA in Oregon in Figure 3-C). As expected, $0/module recycling fees combined with transportation costs of $0.0/t*km makes recycling the most economical pathway (since landfill fees are not null in this scenario) and thus leads to 100% recycling rates in all PCAs (Figure 3-D).

Those results further highlight that recycling economics are complicated since both the process and logistic costs of recycling would need to beat those of landfilling. However, economics and regulations may not be the only factors affecting PV EOL management behaviors (Walzberg, Carpenter, and Heath 2021). The next section presents results of the CE ABM when only the PBC is accounted for with results when all the TPB factors (i.e., social and economic) are included.

Results under different assumptions on the socio-economic factors affecting recycling

In Figure 4, results from the CE ABM under different assumptions regarding how PV EOL management decisions are made are presented. As explained in Walzberg, Carpenter, and Heath (2021), early adopters and positive attitudes towards recycling from PV owners could boost recycling – even if the economics are not favorable (Figure 4-B). On the contrary, if PV EOL management decisions are purely based on costs, recycling is not likely to occur – at least if landfill costs and proximity do not significantly change from the current situation. In this comparison, a recycling cost of $20/module is assumed, which is within the range usually reported by the industry (Curtis, Buchanan, Heath, et al. 2021; Curtis, Buchanan, Smith, et al. 2021).

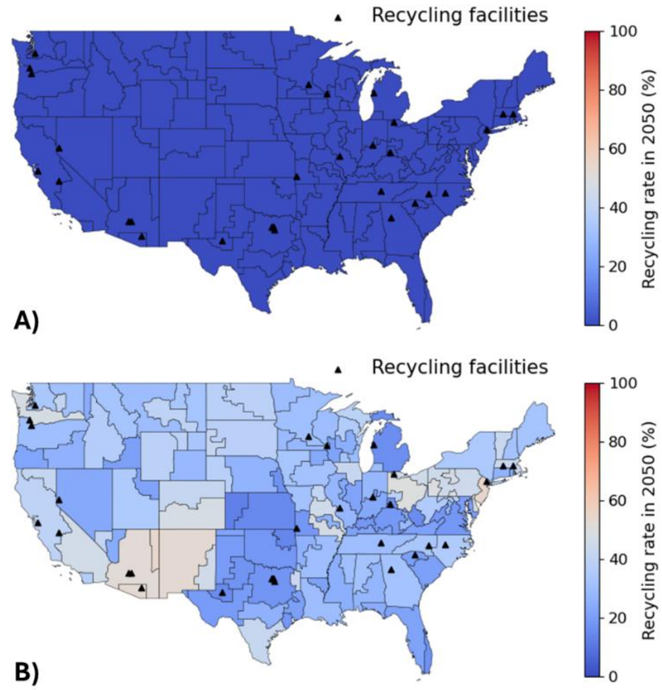

Figure 4: Cumulative recycling rate in 2050 across the U.S. electricity grid power control areas. In this figure, recycling fees are $20/module, reported individual landfill fees are used, and transportation costs are $1.5/t*km. A) when only the perceived behavioral control (PBC) in the theory of planned behavior (TPB) is used in agent's PV EOL management decisions. B) when all factors in the TPB are used in agent's PV EOL management decisions (PBC, attitude, and subjective norms).

The social factors of the TPB (attitude and subjective norms) increase the overall recycling rate during the 2020-2050 period from 0% (Figure 4-A) to 33% (Figure 4-B). Although knowing exactly all the factors that could affect recycling behaviors is difficult, it is interesting to note that while economics are not favorable currently, many PV owners report that they, or have committed themselves to, recycle their EOL PV panels. Indeed, according to an unpublished survey of the PV industry, the current PV recycling rate could be around 10% (EPRI 2024) – significantly higher than what economics alone would predict. Therefore, striving to account for other factors than economic ones could improve the accuracy of predictions on the future state of PV recycling.

For a circular economy to function, in addition to positive socio-economic conditions on the supply side (leading to material recovery), the demand side also needs to be favorable (to lead to the purchase and use of the recovered materials). The next section discusses the potential volumes of recovered material in an ideal PV recycling scenario and the associated domestic demand for those materials.

Comparing supply and demand of materials recovered form PV recycling

This section first explores two potential markets for recovered silicon, which is currently hampered by lack of high value end markets. The domestic market for silicon carbide (SiC) in electronics is estimated to be approximately 150,000 Mt (requiring about 100,000 Mt of silicon) in 2022. (Estimation based on market valuation and SiC prices (MarketsandMarkets 2022; HSA 2022; Data Bridge Market Research 2022)). By comparison, if all silicon from all EOL PV was recycled during the 2020-2050 period (100% recycling rate), 260,000 Mt of this material would be recovered, that is, less than 10 times the demand from the electronics SiC market over that same period if the demand stays similar to that of 2022. Thus, the small supply from EOL PV may not be worth pursuing by U.S. electronics manufacturers.

The domestic market for silumin (a silicon-aluminum alloy used in the automotive industry) was estimated at around 130,000 Mt (requiring about 13,000 Mt of silicon (Bogdanova, Merkulova, and Gil'manshina 2019)) in 2022. (Estimation based on market valuation and silumin prices (Made-in-China 2023; Skyquest 2021)). By comparison, if all silicon from all EOL PV was recycled during the 2020-2050 period (100% recycling rate), 260,000 Mt of this material would be recovered, about two-thirds of the demand from silumin over that same period if the demand stays similar to that of 2022. On the contrary to SiC, aluminum manufacturers may benefit from using scrap silicon from EOL PV.

Finally, crushed glass from EOL PV could be used to make ground glass pozzolan (GGP) and replace part of the cement in concrete. The domestic market for GGP was estimated at around 18,000,000 Mt in 2017 (based on the market for coal fly ash that GGP would compete with (ACAA 2018)). By comparison, if all glass from all EOL PV was recycled during the 2020-2050 period (100% recycling rate), 5,000,000 Mt of this material would be recovered, over twenty-nine times the demand for coal fly ash over that same period if the demand stays similar to that of 2017. Thus, the small supply from EOL PV may not be worth pursuing by concrete manufacturers, except if other factors such as embodied GHG emissions would be highly valued as additional benefits, or the supply of coal fly ash decreases.

Overall, the comparison between supply and demand for some of the materials that can be recovered from EOL PV shows that it might prove difficult to find applications for them. Using silicon scrap from EOL PV to make silumin seems a promising avenue – albeit if its quality is compatible with silumin manufacturing requirements.

Conclusions & Recommendations

This study strived to analyze different socio-economic conditions that could affect EOL PV management. Given that about 8 million metric tons (Mt) of EOL PV materials are expected to be generated in the next 25 years, building a circular PV industry could enhance the resilience and sustainability of the U.S. economy. This study showed that while the economics of a circular PV industry are not currently favorable given the nascence of the industry, several pathways could help get there. Constraining measures (such as more stringent regulations), enhanced profitability (through optimized reverse supply chains), and leveraging PV stakeholders' commitment to sustainability could all enhance the PV recycling rate.

Specifically, results show that regulations, landfill and recycling costs, and transportation distances could all affect recycling rates. For instance, in the hypothetical and extreme scenario restricting landfilling options (Figure 2-A), the recycling rate is up to 75%. Regarding the correspondence between demand and supply of EOL PV materials, we found that, if Silicon were recovered during recycling, about 260,000 metric tonnes of this material would be obtained between now and 2050. This recovered amount is nearly enough to supply all domestic Silicon demand from silumin manufacturing for that same period. Thus, for the two Silicon end markets investigated, silumin appear to be the most feasible.

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