공학박사 학위논문

Comparative study on environmental impacts of end-of-life photovoltaic panel treatments using life cycle assessment methodology

전과정 평가를 통한 사용 후 태양광 패널 처리 방안에 대한 환경영향 비교 연구

2021 년 8 월

서울대학교 대학원

건설환경공학부

서 보 라

Comparative study on environmental impacts of end-of-life photovoltaic panel treatments using life cycle assessment methodology

지도 교수 김 재 영

이 논문을 공학박사 학위논문으로 제출함

2021년 6월

서울대학교 대학원

건설환경공학부

서 보 라

서보라의 공학박사 학위논문을 인준함

2021년 6월

위 원 장 ______ 남 경 필 ______

부위원장 ______ 김 재 영 ______

위 원 ______ 이 승 희 ______

위 원 ______ 주 문 솔 ______

위 원 ______ 최 정 권 ______

Abstract

Comparative study on environmental impacts of end-of-life photovoltaic panel treatments using life cycle assessment methodology

Bora Seo

Department of Civil and Environmental Engineering

Seoul National University

In June 2020, Siberia experienced abnormally high temperatures with a record-breaking 38℃ in the Arctic Circle. Carbon dioxide (CO_2) is well-known to be a major greenhouse gas (GHG) responsible for global warming. Among several CO_2 emission sources, the contribution of the energy sector accounts for 58.8% of all GHG emissions worldwide. Recently, efforts were undertaken to encourage using renewable energy sources to minimize environmental impacts. Renewable energy sources such as solar energy, wind energy, biomass energy, and geothermal energy provide a promising opportunity for mitigating GHG emission and global warming as substitutes for fossil fuels and could supply 20%–40% of the primary energy demand in 2050.

Among those renewable energy sources, solar energy is the most abundant and accessible, in both direct and indirect pathways. By using photovoltaic (PV) cells, solar energy may be directly converted to electricity without emissions, noise, and vibration. In the last decade, the worldwide PV market has increased rapidly. At the end of 2017, the cumulative global installed PV capacity reached 397.4 GW. The typical lifespan of PV panels is 25–30 years, and PV panel waste would thus become a main environmental issue within the next few years. Countries with high cumulative installation capacity would encounter the largest burden of PV waste in

the future. At the end of 2050, the cumulative global PV waste amounts could reach 60–78 million tons.

Life cycle assessment (LCA) is an effective tool for systematically evaluating the environmental impact of products and/or processes during their life cycle, including manufacturing, operation, and end-of-life (EoL) disposal. Although the environmental impact of PV systems focusing the production and operation phases has been extensively investigated over the past few years, limited studies have focused on the management of EoL PV panels.

Considering the dramatic increase in EoL PV panels as an emerging waste problem in the field of solid waste management, a proper PV waste management protocol has to be established to support a circular economy. Therefore, recovering materials from EoL PV panels is proposed as a primary strategy to reduce the negative environmental impacts, aiding sustainability. The EoL treatment process starts by transporting EoL panels to the recycling facility.

The LCA results of the EoL PV treatment without considering the avoided impacts related to the material recovery demonstrated that the environmental impacts of high-level treatment were higher than those of low-level treatment because of higher environmental burdens associated with additional processes requiring chemicals, electricity, and thermal treatment. When considering environmental disburden associated with material recovery, the net environmental benefits from the high-level treatment were almost two times higher than those of low-level scenarios.

The main empirical findings derived from this study can be summarized as follows. The environmental impacts associated with the material recovery process were assessed for three leaching agents: HNO_3, I_2–KI, and thiourea, to choose the most environmentally friendly alternative. The LCA results for using thiourea showed that the environmental burdens during chemical treatment process were lower than those for the other two leaching agents. However, the net benefit considering material recovery was higher when using HNO_3 as a leaching agent than that of the two other

leaching methods, based on the level of endpoint analysis. Comparing the environmental impacts related to the transportation of PV wastes, the scenario that included transporting the wastes to the recycling facility via the collection center had 14.2% to 26.2% less adverse environmental impact than the scenario that involved directly transporting the wastes to the recycling facility. This reduction in adverse environmental impact was due mainly to weight reduction after pre-treatment and transport efficacy provided by a large-sized vehicle even though the traveling distance via the collection center increased.

When the secondary production of material was considered, some indicators could not exceed the environmental burdens of the recycling processes. Consequently, the findings associated with the treatment of EoL PV panels suggest that resource recovery and transportation should be considered to reduce the economic and environmental burdens.

Keywords: End-of-life management, Life cycle assessment, Photovoltaic panel, Resource recovery, Transportation
Student Number: 2011-30270

Abbreviations

a-Si	Amorphous Silicon
c-Si	Crystalline silicon
CdTe	Cadmium Telluride
CIGS	Copper Indium Gallium Selenide
CRT	cathode-ray tube
EoL	End-of-Life
EPBT	energy payback time
EPR	Extended Producer Responsibility
EU	European Union
EVA	ethylene-vinyl acetate
EVs	Electric vehicles
FDP	Fossil depletion
FEP	Freshwater eutrophication
FETP	Freshwater ecotoxicity
FRELP	Full Recovery EoL Photovoltaics
GHG	Greenhouse gas
GWP	global warming potential
HTP	Human toxicity
IEA	International Energy Agency
IRENA	International Renewable Energy Agency
KEI	Korea Environment Institute
KIER	Korea Institute of Energy Research
KIET	Korea Institute for Industrial Economics and Trade
LCA	Life Cycle Assessment
LCD	liquid crystal display
LCIA	life cycle impact analysis
mc-Si	multi-crystalline Silicon
MDP	Metal depletion
METP	Marine ecotoxicity
MSA	methane sulfonic acid

NEDO	New energy industrial technology development organization
ODP	Ozone depletion
OECD	Organization for Economic Cooperation and Development
PCB	printed circuit board
POFP	Photochemical oxidant formation
PV	photovoltaic
SRMs	secondary raw materials
TAP	Terrestrial acidification
TETP	Terrestrial ecotoxicity
WEEE	waste electrical and electronic equipment

List of figures

List of tables

Table of Contents

Chapter 1. Introduction

1.1. Background

In July 2020, Siberia experienced abnormally high temperatures with a record-breaking 38℃ inside the Arctic Circle (Sherwood, 2020). Evidence indicates that the global climate is rapidly changing in relation with the use of fossil fuels. Carbon dioxide (CO_2) is a well-known major greenhouse gas (GHG) responsible for global warming. Among several CO_2 emission sources, the contribution of the energy sector accounts for 58.8% of all GHG emissions worldwide (IEA, 2011). Thus, efforts have been made to encourage using renewable energy sources to minimize environmental impacts. Renewable energy sources, such as solar energy, wind energy, biomass energy, and geothermal energy, provide a promising opportunity for mitigating GHG emissions and global warming by substituting fossil fuels. Renewables are expected to supply 20%–40% of the primary energy requirement in 2050 (Fridleifsson, 2003).

Among those renewable energy sources, solar energy is directly and indirectly the most abundant and accessible. By using photovoltaic (PV) cells, solar energy is directly converted to electricity without emissions, noise, and vibration (Panwar et al., 2011). Currently, silicon-based technology dominates 90% of the PV market, including single crystal (c-Si) and multi-crystalline silicon (mc-Si), commonly referred to as first-generation technology. Second-generation PV technology includes amorphous silicon (a-Si), cadmium telluride (CdTe), and copper indium gallium selenide (CIGS), which use cheap semiconductor thin films. However, the scale-up of CdTe and CIGS is limited because of the efficiency gap between the lab and practical fields, which is ascribed to poor reproducibility and uniformity (Bagnall and Boreland, 2008). Moreover, second-generation PV cells comprise more toxic and harmful materials than first-generation PV cells, such as cadmium and selenium. Several PV technologies are considered third generation, including dye-sensitized, organic, perovskite, and hybrid PV cells. These PV technologies should offer several benefits over previous technologies, including using cheap materials, superior performance, durability, flexibility, and weight reduction (Charles et al., 2016). When considering sustainability, environmental impacts related to the production, transportation, maintenance, and substitution of primary resources should be minimized via recycling.

In the last decade, the worldwide PV market has increased rapidly. At the end of 2017, the cumulative global installed PV capacity reached 397.4 GW (IRENA, 2018). The typical lifespan of PV panels is 25–30 years; therefore, PV panel waste will become the main environmental issue within the next few years. International Renewable Energy Agency (IRENA) first estimated the global PV waste volume until 2050 under two scenarios: regular- and early-loss. Regular-loss assumes a 30-year lifetime of solar panels with no early-loss, whereas early-loss considers infant, midlife, and attrition failures before the 30-year lifespan (IRENA, 2016). Countries with high cumulative installation capacity and/or the most challenging PV targets would encounter the largest PV waste burden in the future. At the end of 2050, the cumulative global PV waste amounts should reach 5.5–6 million tons, which are almost identical to the mass of new installations. From the IRENA report (2016), China, Germany, and Japan are the top three countries with the largest projected cumulative PV waste in 2030. At the end of 2050, China should still have the largest cumulative PV waste amounts, but Germany is set to be replaced by the US, followed by Japan and India. In the Republic of Korea, two scenarios were proposed to estimate the local PV panel waste production (KEIT, 2017; KEI, 2018). In 2020, both scenarios were forecasted to have similar cumulative PV panel waste amounts because most PV panels did not reach their average lifespan. At the end of 2040, however, the difference becomes more pronounced, which could be caused by considering additional loss because of the wide variation in PV panels' average lifetime.

The management of the exponentially increasing number of end-of-life (EoL) PV panels is critical and challenging because they impose serious and crucial environmental concerns. Unfortunately, most member countries of Organisation for Economic Cooperation and Development (OECD) have not yet established and adopted specific strategic frameworks and policies targeting EoL PV waste management (Mahmoudi et al., 2021). The European Union (EU) has established regulatory frameworks based on the Waste Electrical and Electronic Equipment (WEEE) Directive, including EoL PV waste management and recycling technologies for material recovery. This directive has set rules to ensure that PV manufactures have liability for the collection, handling, and treatment costs while satisfying the WEEE Directive's requirements and responsibilities (Sharma et al., 2019). The Korea Ministry of Environment has also provided a legislative notice on the Act on Resource Circulation of Electrical and Electronic Equipment and Vehicles by adopting extended producer responsibility (EPR) to 23 subjects, including PV panel waste (ME, 2019). This directive will only be effective from 2023, after laying the foundation for recovery and storage strategies and appropriate technologies for recycling PV waste until 2022. This subject extension benchmarked the EU framework.

Life cycle assessment (LCA) is an effective tool for the systematic evaluation of the environmental impact of products and/or processes during their life cycles, including manufacturing, operation, and EoL disposal (Hauschild and Huijbregts, 2015; IEA, 2020; Wäger and Hischier, 2015). Although the environmental impact of PV systems with a focus on the production and operation phases has been investigated extensively over the past few years, only a few studies have focused on the management of the EoL of PV panels (Bracquene et al., 2018; Vellini et al., 2017; Huang et al., 2017). As opposed to disposal in landfills, proper recycling of EoL PV panels is economically and environmentally favorable because they contain both valuable and hazardous materials. Corcelli et al. (2018) reported that the recycling processes were attractive in many impact categories, such as human toxicity, freshwater eutrophication, and fossil depletion indicators. Particularly, the environmental benefits are primarily linked to recovering aluminum, silicon, and copper by replacing primary raw materials with equivalent recycled ones. Efficient EoL recycling can also considerably reduce energy consumption and GHG emissions (Deng et al., 2019).

1.2. Objectives

Considering that a dramatic increase in EoL PV panels is an emerging problem in solid waste management, a proper PV waste management protocol must be established to support circular economy. EoL PV panels cause several significant environmental burdens if not properly treated (Mahmoudi et al., 2019). The recycling of decommissioned PV panels can reduce the significant increase in global demand for raw materials. More importantly, it can be expected to minimize several negative environmental impacts when it is compared with the conventional landfill approach. Ardente et al. (2019) reported that EoL PV panel recycling can reduce the environmental impacts considerably and save precious materials. For example, the process of extracting raw materials requires high energy to manufacture PV panels, and CO_2 emissions are produced. One of the key factors that affects the environmental results of LCA is the transportation burden (Xu et al., 2013). A recent study demonstrated that transporting the EoL PV panels to the recycling site mainly governed all the considered impact categories during the LCA study of the Full Recovery End of Life Photovoltaics (FRELP) recycling process (Latunussa et al., 2016).

Among the stages of manufacturing, operation, and EoL of PV panels, comparatively fewer studies have been made for the management of EoL PV panels in terms of environmental impacts. According to IEA Methodology guidelines (2020), the system boundaries at the EoL stages include (1) deconstruction and dismantling, (2) transport, (3) processing the waste, (4) recycling and reuse, and (5) disposal. LCA has been applied extensively to the recycling and reuse of EoL PV panels, but only a few studies have dealt with the transport of PV wastes.

From previous studies, the recycling process of PV modules can be broadly divided into (1) delamination, (2) material separation, and (3) material extraction and purification (Maani et al., 2020; Lunardi et al., 2018; Tao and Yu, 2015). For delamination, thermal treatment has been widely adopted because of its lower impact than methods such as chemical, electro-thermal heating, and pyrolysis. Recently, many studies have been conducted to recover valuable materials, such as glass, silicon, aluminum, copper, and silver, during PV module recycling. Although the collection, transfer, and transport of waste are critical activities in the waste management system, the environmental impacts associated with PV waste transport have rarely been addressed. Therefore, it can be postulated that EoL PV panel material recovery and their transportation are critical in determining the environmental impact during the recycling process. Figure 1.1 shows the full PV module lifecycle, and the dotted-line box indicates the main scope of this study. The detailed objectives of the research are as follows:

(i) to compare the two EoL scenarios of c-Si PV panels considering environmental impacts; high and low levels represent the most and least material recovery, respectively.

(ii) to investigate the applicability and environmental benefits of alternative resource recovery technologies by substituting conventional leaching techniques.

(iii) to evaluate environmental impacts driven by EoL PV panel transportation based on the proposed EoL scenarios and travel conditions in the Republic of Korea.

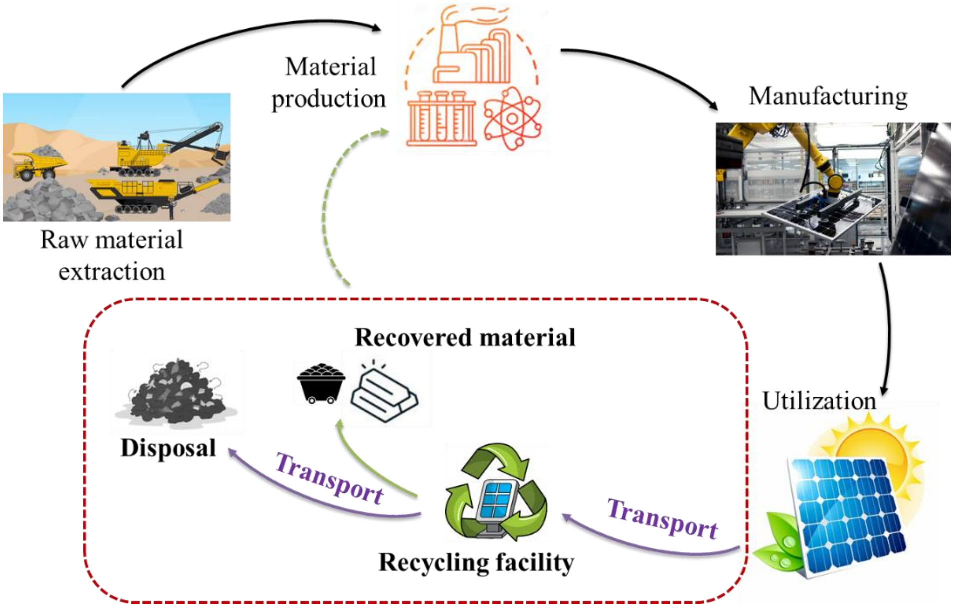


Fig. 1.1. Full life cycle of a PV module: the dotted-line box indicates the main scope of this study.

References

Bagnall, D.M., Boreland, M., 2008. Photovoltaic technologies. Energy Policy 36, 4390–4396.

Bracquene, E., Peeters, J.R., Dewulf, W., Duflou, J.R., 2018. Taking Evolution into Account in a Parametric LCA Model for PV Panels. Procedia CIRP 69, 389–394.

Charles, R.G., Davies, M.L., Douglas, P., 2017. Third generation photovoltaics-Early intervention for circular economy and a sustainable future. 2016 Electron. Goes Green 2016+, EGG 2016 1–8.

Corcelli, F., Ripa, M., Leccisi, E., Cigolotti, V., Fiandra, V., Graditi, G., Sannino, L., Tammaro, M., Ulgiati, S., 2018. Sustainable urban electricity supply chain – Indicators of material recovery and energy savings from crystalline silicon photovoltaic panels end-of-life. Ecol. Indic. 94, 37–51.

Deng, R., Chang, N.L., Ouyang, Z., Chong, C.M., 2019. A techno-economic review of silicon photovoltaic module recycling. Renew. Sustain. Energy Rev. 109, 532–550.

Fridleifsson, I.B., 2003. Status of geothermal energy amongst the world's energy sources. Geothermics 32, 379–388.

Hauschild, M.Z., Huijbregts, M.A.J. (Eds.), 2015. Life cycle impact assessment. Springer, Netherlands, Dordrecht.

Huang, B., Zhao, J., Chai, J., Xue, B., Zhao, F., Wang, X., 2017. Environmental influence assessment of China's multi-crystalline silicon (multi-Si) photovoltaic modules considering recycling process. Sol. Energy 143, 132–141.

International Energy Agency (IEA), 2011, IEA Statistics 2011 Edition: CO_2 emissions from fuel combustion highlights. International Energy Agency, Paris.

International Energy Agency (IEA), 2020. Methodology guidelines on Life Cycle Assessment of photovoltaic

International Renewable Energy Agency (IRENA), 2018. Renewable power generation costs in 2017 - Key findings and executive summary. Int. Renew. Energy Agency 16.

International Renewable Energy Agency (IRENA), 2016. End-Of-Life management: solar photovoltaic panels.

Korea Institute for Industrial Economics and Trade (KIET), 2017. Revitalization of renewable energy industry through post-management system development

Korea Environment Institute (KEI), 2018. Management status and improvement plans of waste solar panels

Latunussa, C.E.L., Ardente, F., Blengini, G.A., Mancini, L., 2016. Life Cycle Assessment of an innovative recycling process for crystalline silicon photovoltaic c. Sol. Energy Mater. Sol. Cells 156, 101–111.

Lunardi, M., Alvarez-Gaitan, J., Bilbao, J., Corkish, R., 2018. Comparative Life Cycle Assessment of End-of-Life Silicon Solar Photovoltaic Modules. Appl. Sci. 8, 1396.

Maani, T., Celik, I., Heben, M.J., Ellingson, R.J., Apul, D., 2020. Environmental impacts of recycling crystalline silicon (c-Si) and cadmium telluride (CDTE) solar panels. Sci. Total Environ. 735, 138827.

Mahmoudi, S., Huda, N., Alavi, Z., Islam, M.T., Behnia, M., 2019. End-of-life photovoltaic modules: A systematic quantitative literature review. Resour. Conserv. Recycl. 146, 1–16.

Mahmoudi, S., Huda, N., Behnia, M., 2021. Critical assessment of renewable energy waste generation in OECD countries: Decommissioned PV panels. Resour. Conserv. Recycl. 164, 105145.

Ministry of Environment (ME), 2019. Act on resource circulation of electrical and electronic equipment and vehicles. Republic of Korea.

Panwar, N.L., Kaushik, S.C., Kothari, S., 2011. Role of renewable energy sources in environmental protection: A review. Renew. Sustain. Energy Rev. 15, 1513–1524.

Sharma, A., Pandey, S., Kolhe, M., 2019. Global review of policies & guidelines for recycling of solar pv modules. Int. J. Smart Grid Clean Energy 8, 597–610.

Sherwood, S.C., 2020. Adapting to the challenges of warming. Science (80-.). 370, 782–783.

Tao, J., Yu, S., 2015. Review on feasible recycling pathways and technologies of solar photovoltaic modules. Sol. Energy Mater. Sol. Cells 141, 108–124.

Vellini, M., Gambini, M., Prattella, V., 2017. Environmental impacts of PV technology throughout the life cycle: Importance of the end-of-life management for Si-panels and CdTe-panels. Energy 138, 1099–1111.

Wäger, P.A., Hischier, R., 2015. Life cycle assessment of post-consumer plastics production from waste electrical and electronic equipment (WEEE) treatment residues in a Central European plastics recycling plant. Sci. Total Environ. 529, 158–167.

Xu, Q., Yu, M., Kendall, A., He, W., Li, G., Schoenung, J.M., 2013. Environmental and economic evaluation of cathode ray tube (CRT) funnel glass waste management options in the United States. Resour. Conserv. Recycl. 78, 92–104.

Chapter 2. Literature review

2.1. Introduction

Increasing the renewable energy share in the total energy mix will not only help overcome the dependence on limited fossil fuels but also undoubtedly contribute to increased energy security and a reduction in global warming and air pollution problems. Renewable energy plays a crucial role in the sustainable development of future energy strategies (Weitemeyer et al., 2015; Ozoemena et al., 2016). The most common renewable energy sources are solar, wind, hydro, geothermal, and biomass (IRENA, 2020). Among these sources, solar energy is considered to have a high potential for significant electricity generation via photovoltaic (PV) conversion because of its abundance and worldwide accessibility (Dale and Benson, 2013; Liu et al., 2017).

The solar energy sector is one of the fastest-growing energy sectors worldwide, with a growth rate of 35–40% per year (Tyagi et al., 2013). The year 2019 became another historic year for solar energy because global installed power capacity had reached approximately 600 GWp (Fraunhofer ISE, 2020). This global installed PV capacity in 2019 was almost six times larger than that in 2012. Considering the lifespan of PV panels to be approximately 25–30 years, PV waste will soon proliferate dramatically in response to the remarkable growth rates. The amount of cumulative PV waste was estimated as 250,000 tons at the end of 2016, while it is estimated that there will be 78 million tons by 2050 worldwide according to the early loss scenario of International Renewable Energy Agency (IRENA, 2016). It is expected that this massive waste will be a global burden to the environment and the economy.

Various reclaimable resources in PV waste, such as silicon (Si), glass, aluminum (Al), copper (Cu), and silver (Ag), will likely be of significant importance as underground resources further deplete. Furthermore, some types of PV panels such as CdTe or (lead-based) perovskite pose a significant environmental threat when they are not properly disposed of. In most countries, adequate regulations for the recycling process of end-of-life (EoL) PV panels have not been fully established. The European Union (EU) has adopted PV waste regulations, including collection, recovery, and recycling of EoL PV panels. Since 2014, decommissioned PV panels have been included in the list of the EU Waste Electrical and Electronic Equipment (WEEE) directive based on the extended-producer-responsibility (EPR) principle (EC, 2012). Depending on the different EoL scenarios, up to 820,000 tons of discarded EoL PV panels are expected to be discarded in Korea by 2040 (KEI, 2018). Therefore, the Korea Ministry of Environment has provided a legislative notice on the Act on Resource Circulation of Electrical and Electronic Equipment and Vehicles with the adoption of EPR to 23 subjects, including PV panel waste (ME, 2019). This directive will only be effective from 2023, after laying the foundation for recovery and storage strategies and appropriate technologies for recycling PV waste until 2022. This extension of the subjects benchmarked the EU framework.

Life cycle assessment (LCA), based on the ISO 14040 standard (ISO, 2006) is an effective tool for systematically assessing the environmental impact of products or processes during their life cycle, including manufacturing, operation, and EoL disposal (Hauschild and Huijbregts, 2015). Although the environmental impact of PV systems with a focus on the production and operation phases has been extensively investigated over the past few years, limited studies have explored in depth the management of EoL PV panels (Huang et al., 2017; Vellini et al. 2017; Bracquene et al., 2018). As opposed to disposal in landfills, proper recycling of EoL PV panels is economically and environmentally favorable because they contain both valuable and hazardous materials. Corcelli et al. (2018) reported that the recycling processes were attractive in many impact categories such as human toxicity, freshwater eutrophication, and fossil depletion indicators. In particular, the environmental benefits are primarily linked to the recovery of aluminum, silicon, copper, etc., by replacing primary raw materials with equivalent recycled ones. Efficient EoL recycling can also considerably reduce energy consumption and greenhouse gas (GHG) emissions (Deng et al., 2019).

There are many types of solar cells that can be broadly grouped into crystalline silicon (c-Si), thin-film, and third-generation PV panels that utilize emerging technologies, such as concentrating, plastic, and dye-sensitized PV (Paiano, 2015). Among these, c-Si PV is the most common technology, accounting for 94.5% of the world market as of 2019, while the global market share of thin-film is approximately 5.5% (Fraunhofer ISE, 2020). By the end of 2030, the market share of c-Si PV is expected to decrease to 44.8%, while that of third-generation technologies is expected to reach 44.2% (IRENA, 2016). Hence, for the next three decades, the recycling of c-Si PV panels is the most important PV-related environmental challenge worldwide. This chapter attempts to (1) outline the forecast of the global status of renewable energy consumption and PV waste generation worldwide under different scenarios, (2) present and summarize recent literature on recycling technologies with a focus on reclaimable resources from c-Si PV panel, and (3) evaluate LCA results for EoL c-Si PV panels. It is expected that the tremendous amount of c-Si PV panels can cause serious environmental and economic impacts in the coming decades unless treated and managed appropriately (i.e., recycled) (Huang et al., 2017; Deng et al., 2017). This chapter scrutinizes the LCA studies for EoL c-Si PV panels with special attention to the environmental benefits associated with the recycling of materials to identify challenges for future research in this area.

2.2. Solar photovoltaic energy as an emerging renewable energy source

2.2.1. Global status and the market for renewable energy

According to the latest data released by the International Renewable Energy Agency (IRENA, 2020), global renewable energy capacity continues to develop rapidly. As shown in Fig. 2.1, the global renewable energy capacity increased by 175,791 MW and reached 2,537 GW by the end of 2019, representing a 7.4% increase compared with 2018. Since 2010, growth in the global capacity has been remarkably boosted by technological improvements and favorable policy environments worldwide. Among renewable energy sources, such as hydropower, wind, solar, and biomass, solar and wind power systems have grown steadily over the past decade, mainly due to substantial reductions in capital and generation costs (Devabhaktuni et al., 2013). Between 2010 and 2019, the growth rate of solar and wind power capacity increased by 1,311% and 244%, respectively. This considerable growth rate is attributable to the fact that solar energy is inexhaustible, it is available almost worldwide, and solar power systems are easy to install and maintain (Lewis, 2007; Weitemeyer et al., 2015).

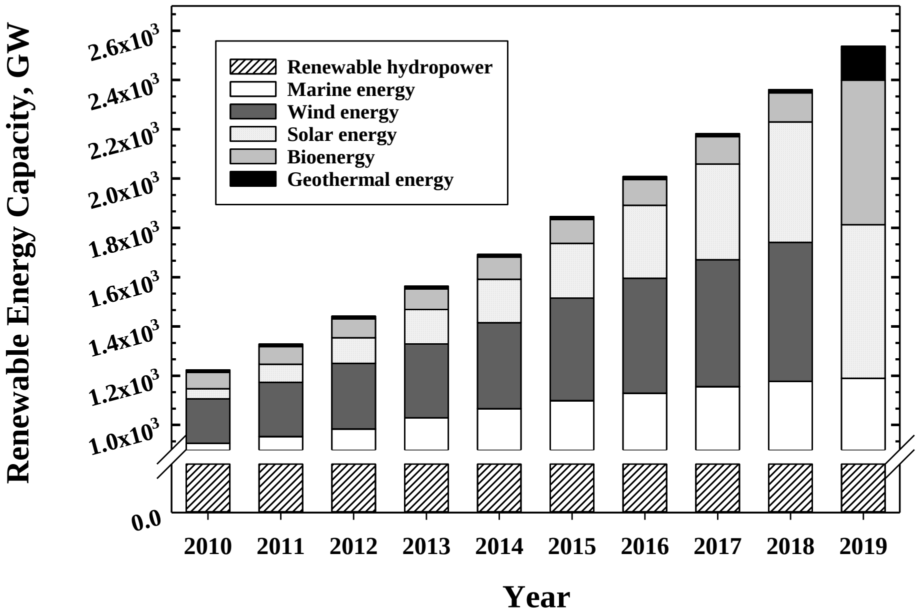


Fig. 2.1. Total renewable energy capacity between 2010 and 2019, worldwide. Source: IRENA, 2020.

Consequentially, a large number of PV panels and wind turbines have been installed worldwide during the past decade. As shown in Fig. 2.2, IEA (2019) and IRENA (2019) project that solar and wind power will dominate electricity production, each with a share of approximately 21% in 2040. Though the projected changes are slightly different, the trends are quite similar, indicating that solar and wind power are expected to increase more rapidly than other renewable energy sources in the next 30 years. Undoubtedly, this enormous share results from a wide geographical distribution as well as the huge energy conversion potential (Shi et al., 2020). However, such an exponential growth may pose a major environmental challenge due to the production of new types of waste streams (e.g., PV and wind turbine blade wastes) in the coming decades.

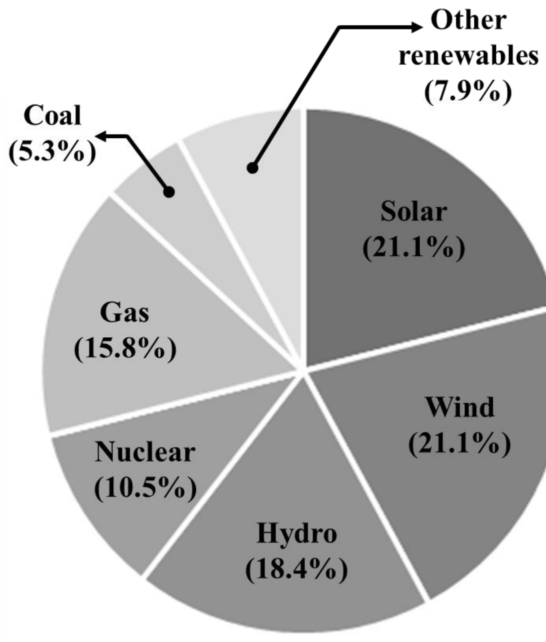


Fig. 2.2. Estimation of share of global electricity production in 2040 (from sustainable development scenario of IEA, 2019).

In the last decade, the global solar PV market has experienced rapid expansion. As shown in Fig. 2.3, China is leading the global PV markets for both cumulative and installed capacities in 2019. Indeed, China installed 30 GW of solar energy in 2019, and its cumulative installed capacity reached 204.7 GW at the end of 2019. Vietnam and Ukraine are emerging markets with 4.8 and 3.5 GW of the annual installed capacity of solar electricity in 2019, respectively. At the end of 2019, the cumulative capacity of solar PV from the top 10 countries reached 627 GW, accounting for 24.7% of the global renewable energy capacity.

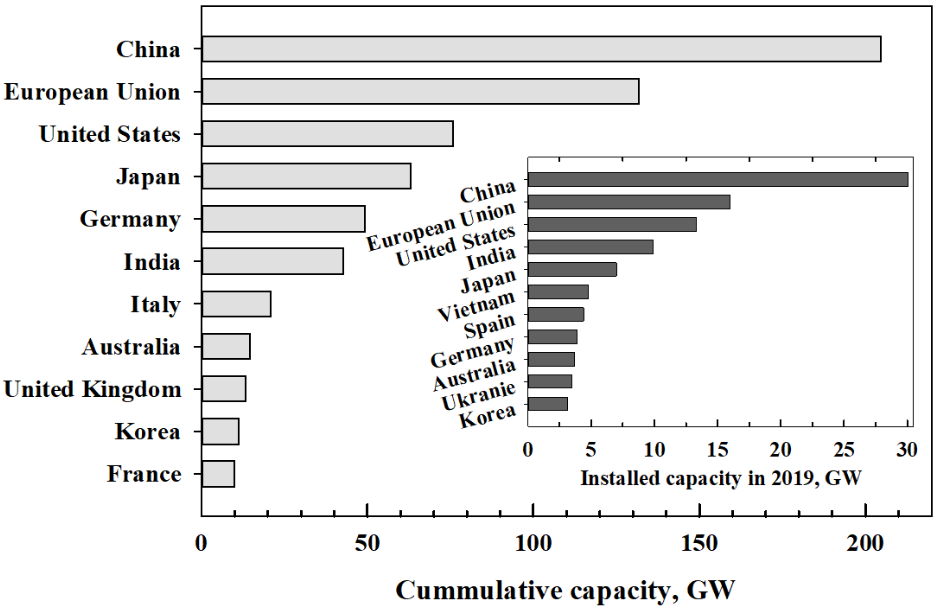


Fig. 2.3. Cumulative and total installed (the inset figure) capacities of the top ten photovoltaic (PV) markets in 2019. Sourced from IEA PVPS, 2020.

2.2.2. Status and the market for renewable energy in Korea

The evolution of renewable energy capacity in the Republic of Korea (Korea) is analogous to global trends. Data collected between 2010 and 2019 show that in Korea, the wind power capacity increased linearly by a factor of 4 and 16 for the wind and solar power capacity, respectively (Fig. 2.4). Solar power is expected to be used more frequently in the future (Chung and Kim, 2018). According to the “Renewable Energy 2030 Implementation Plan” (MOTIE, 2017), the Korean government is planning to increase the cumulative installed capacity of renewable energy to 63.8 GW by 2030. Moreover, the Korean government plans to increase the percentage contribution of solar energy among all the renewable energy sources, representing up to 63% of renewable capacity by 2030.

Since the Korean government aims to reduce national greenhouse gas emission levels by 37% relative to the “business as usual” scenario (i.e., 850.6 to 535.9 million ton) by 2030, a new growth opportunity of the Korean market has emerged. In fact, the overall installed capacity increased to 11.2 GW with 3.1 GW installed in 2019.

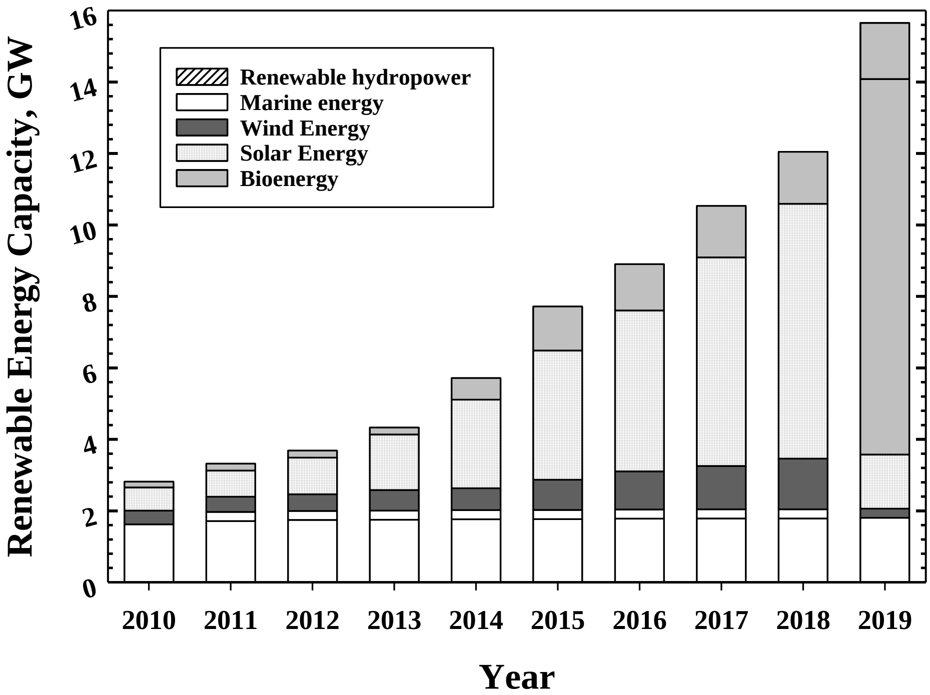


Fig. 2.4. Total renewable energy capacity between 2010 and 2019 in the Republic of Korea. Source: IRENA, 2020.

2.2.3. Global projection of PV waste generation

With the massive growth in solar PV utilization in recent years and the expected dramatic increase in demand by 2050, PV panel waste has become a major issue of concern worldwide. When converting solar energy to electricity, there are irreversible losses associated with environmental factors such as high temperature, relative humidity, and panel soiling (Ross, 2014; Flowers et al., 2016). The average degradation rate, the rate at which solar panels lose efficiency over time, is estimated to be in the range of 0.5% to 0.8% per year (Jordan and Kurtz, 2013). Based on this estimation, a life expectancy of 25–30 years is expected for well-maintained PV systems (IEA, 2009; Perpiñan et al., 2009).

Fig. 2.5 depicts the predicted cumulative waste volumes of EoL PV panels generated by five countries in 2020 and 2030 based on two different scenarios. The cumulative global waste volumes of the EoL PV panels can be estimated based on the Weibull function showing different shape factors for regular and early loss scenarios. Input assumptions of the regular loss scenario are the 30-year average panel lifetime and 99.99% probability of loss after 40 years, while the early loss scenario additionally considers infant, midlife, and wear out failures before its lifespan (IRENA, 2016). Simulations as shown in Fig.2.5 projects Germany as the largest producer of PV waste in 2020. However, the amount of PV waste from China dominates the largest portion in terms of early loss scenario at the end of 2030, while Germany still holds the first place in terms of the regular loss scenario. This indicates that there has been a dramatic increase in newly installed PV panels in China since the late 2010s.

Indeed, China globally led the PV market and solar power production in 2019; it accounted for 28% of the gross electricity generation worldwide (IEA, 2020).

Many other researchers have estimated the amount of PV waste based on the two different scenarios, regular- and early-loss, and the Weibull function with a lifespan of 30 years (Santos and Alonso-García and Sica et al.). Santos and Alonso-García (2018) reported that the cumulative PV waste would reach 100,000 tons by 2030 because of the booming of PV installation between 2007 and 2008 in Spain. On the other hand, Paiano (2015) employed the simplest method assuming the installed PV panels become waste after 25 years, the average lifetime of PV technology; the time shifting of 25 years from the years of installation to waste generation. Recently, the importance of sensitivity analysis has been pointed out when forecasting the stream of PV waste. Peeters and his colleagues (2017) asserted that the uncertainty would be considered in projecting the amount of PV waste because of the rapid development of PV technology, including the individual materials and components.

PV waste can be framed not only as an environmental and human hazard but also as a potentially valuable resource. PV waste contains a number of hazardous and valuable constituents such as silver, copper, aluminum, and silicon. According to IRENA's report (2016), the cumulative value of raw materials from PV panels can reach up to 450 million US dollars by 2030. Unfortunately, a substantial volume of toxic chemicals is also used to achieve high yield recovery of those materials, which thus requires environmental-friendly recycling strategies. The Basel convention effective from 1992 is a multilateral environmental agreement with a specific aim to regulate the movements of hazardous waste between nations, from developed to less developed countries lacking PV and other e-waste recycling facilities and specific laws and/or focal regulations for the management of PV waste. For example, the improper method of recovering valuables from e-wastes have caused serious health risk and environmental problems with severe land and water pollution in China (Sauser et al., 2014). Therefore, the ethical and technological issues associated with the management of PV waste in developing countries are carefully addressed.

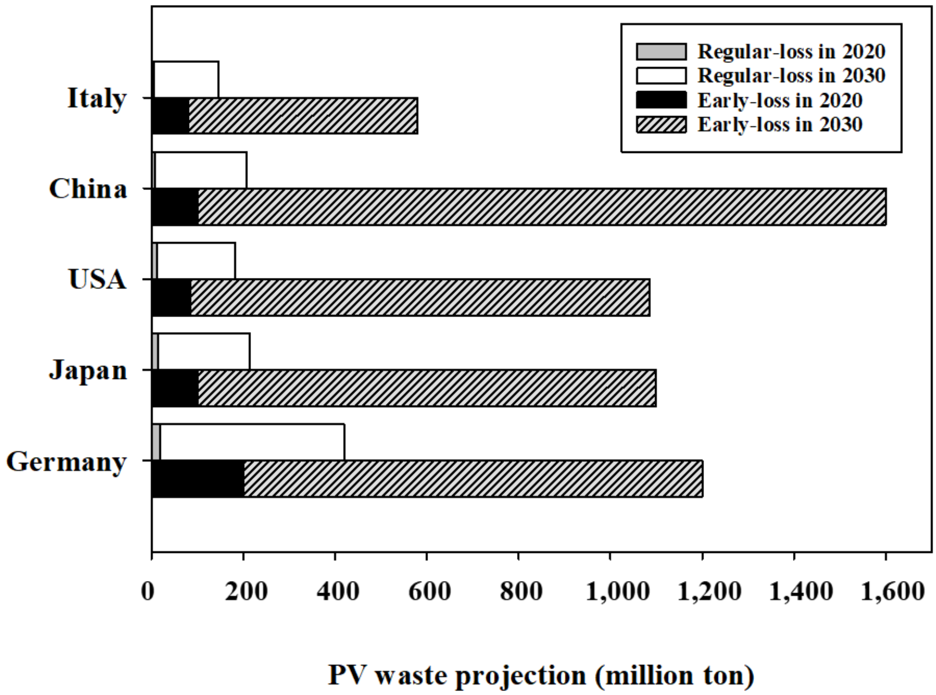


Fig. 2.5. Estimated cumulative waste volumes of end-of-life (EoL) photovoltaic (PV) panels based on early and regular loss scenarios by five countries in 2020 and 2030 (Data source: IRENA, 2016).

2.2.4. Projection of PV waste generation in Korea

Conventionally, the projection of waste PV panels is calculated by considering three assumptions: supply status, average lifetime, and specifications of PV panels. In the Republic of Korea, two different scenarios were suggested to estimate the local production of PV panel waste. In scenario 1, based on the supply status information from the Korea Energy Agency, the amount of waste panels was estimated assuming an average lifetime of 25 years, weight-to-power ratio of 80 ton/MW, and 1% loss of supplied PV panels within 10 years due to unintended causes such as natural disasters (KIET, 2017). These assumptions are somewhat similar to the regular loss scenario of IRENA (2016). In contrast, scenario 2 reflected the “Renewable energy 3020 implementation plan” (MOTIE, 2017) and the projected amount of waste PV panels (KEI, 2018). In this case, the weight-to-power ratio was assumed to be 100 ton/MW, and the average lifetime of PV panels ranged from 15 to 30 years depending on the installation year to reflect the technological improvements of PV panels (KIER, 2015).

Fig. 2.6 depicts the cumulative estimation of PV panel waste by different scenarios over time in the Republic of Korea. In 2020, scenarios 1 and 2 showed the cumulative PV panel waste of 482 and 620 tons, respectively. The difference between the two scenarios was negligible because most PV panels did not reach their average life span. However, the difference between the two scenarios gradually increased until 2040; approximately 288,000 and 820,000 tons were produced according to scenarios 1 and 2, respectively. Compared to Scenario 1, Scenario 2 considers additional loss due to wide variation in the average lifetime of PV panels. Although different numerical assumptions were made, the gradually increasing trend observed in scenarios 1 and 2 was in good agreement with scenarios 4 and 5, which was the estimations driven by the regular and early loss scenarios of IRENA (2016), respectively. In particular, scenario2 showed no significant difference from scenario 3 calculated by assuming the lifetime of PV as 25 years.

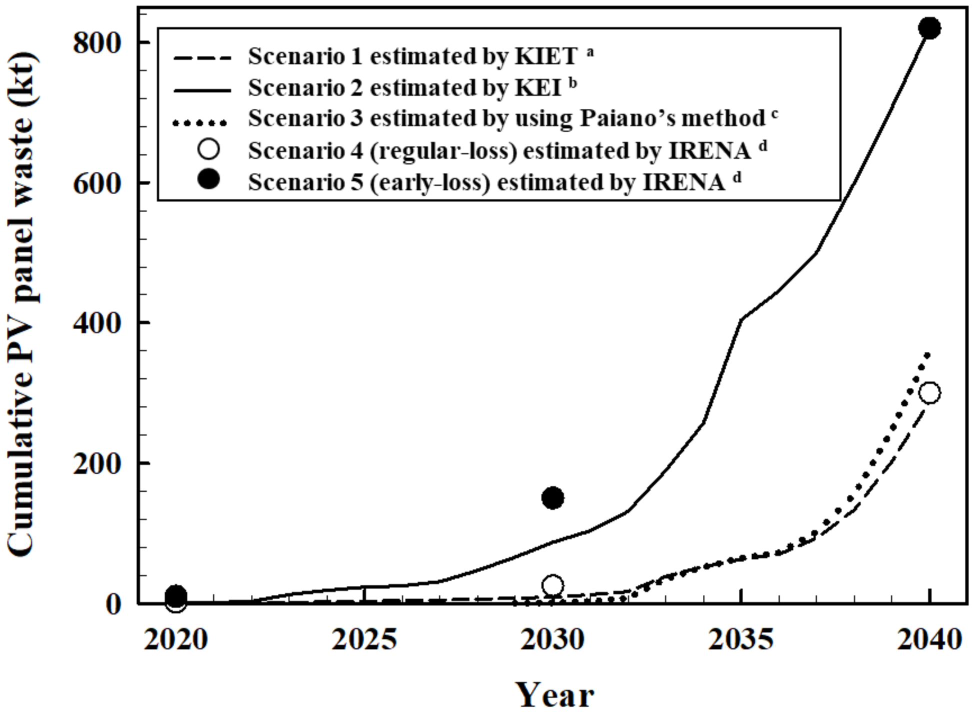


Fig. 2.6. Cumulative estimation of photovoltaic (PV) panel waste by different scenarios until 2040 in the Republic of Korea ([a] KIET, 2017; [b] KEI, 2018; [c] Paiano, 2015; [d] IRENA, 2016).

2.3. c-Si PV panel recycling processes and reclaimable resources

Various processes have been proposed for PV waste recycling; the typical c-Si PV panel recycling process involves (1) Separation of module, (2) EVA removal, and (3) resource recovery, where each sub-process can employ either physical, chemical, thermal, cryogenic, electrical, or combined methods as follow (Deng et al., 2019): First, the aluminum frame is dismantled, and then the cable and junction box are removed from the PV sandwich (i.e., solar cell and back sheet). Subsequently, ethylene vinyl acetate (EVA), which holds the adjacent layers tightly, is removed via thermal, chemical, mechanical, cryogenic, or combined treatment to separate the glass, solar cell, and plastic back sheet. Finally, precious metals such as silver, aluminum, and copper are recovered by hydrometallurgical and electrochemical processes. Table 2.1 lists several representative processes with reclaimable resources in each step. Note that the entire process could be divided by module separation and resource recovery from the cell, which is also referred to as "low-rate" and "high-rate" recycling, respectively (Corcelli et al., 2018). In this section, PV recycling processes are reviewed from the perspective of recycling (i.e., mechanical separation, thermal and electrochemical treatment), and reclaimable resources at each stage are discussed.

2.3.1. Module separation

After transporting the waste PV materials to the recycling facility, the first step is to physically dismantle the aluminum frame and separate the glass, cable, and junction box from the PV sandwich (i.e., solar cell and back sheet layers). This is usually conducted using an electric powered machine, and depending on the method, glass can be recovered either as a crushed form of cullet, or as whole glass using a heated knife (NEDO), or specially developed optical technologies (Loser Chemie). Glass takes almost 70% (weight basis) of PV waste and is specially produced with low iron content (< 0.02%) for exceptional transparency, which has much higher value than the normal glass when recycled. Dismantled aluminum frames and copper wires can also be directly reused or sent to raw material producing facilities depending on their condition. Plastics from the cable and junction box are subjected to a subsequent thermal process (i.e., incineration) for heat recovery or disposed of in landfills.

2.3.2. EVA removal

Subsequently, EVA is removed to separate the glass, solar cell, and plastic back sheet layers from each other. This is usually done using a special organic solvent or by thermal treatment. Crushed PV scraps are subjected to incineration to decompose the polymeric EVA and plastic back sheet, and the resulting heats can be recovered together with residual cell scraps (FRELP). A step-wise increase in temperature (20 to 600 °C) was also proposed for effective decomposition of the polymeric layer (Corcelli et al., 2018). However, emissions from incinerating the halogenated plastic back sheet (fluorine content up to 9%) involve the release of toxic pollutants such as HF (Ardente et al., 2019). Alternative processes, such as pyrolysis followed by solvent dissolution (ELSi) and hot knife (NEDO), were also applied. As waste becomes a major issue, an easy delamination process needs to be considered at the design stage (Latunussa et al., 2016).

2.3.3. Precious metal recovery

With the depletion of underground resources becoming a reality, the concept of urban mining is proposed with respect to e-waste such as PCB, LCD, CRT, and batteries (Cossu and William, 2015). Although solar cells contain minimal amounts (varying from 0.5% to 5% in wt.) of precious metals such as silver and aluminum, the value of recycling them cannot be ignored as the unit prices are high compared to the other reclaimable resource from c-Si PV wastes (Peeter et al., 2017; Kuczyńska-Łażewska et al., 2018). Hydrometallurgical recovery of these metals from PV cells usually involves strong solvents such as cyanides or nitric acid, resulting in the emission of chemicals to air and water, which could be a significant threat to the environment and public health (Latunussa et al., 2016). Recently, an alternative leaching reagent has been proposed using environmentally friendly, nontoxic substitutes for acid leaching (i.e., so-called "green leaching") such as methane sulfonic acid (MSA) (Palitzsch et al., 2014; Yang et al., 2017) and iodine-iodide solution (Chung et al., 2021). In addition to dissolved metals, an insoluble Si wafer is recovered in a mixed metal solution after leaching. An additional hydrometallurgical refining process was conducted to recover pure silicon ingots (more than 99.9999%) (Loser Chemie). Regaining the precious metals as a pure solid form is conducted by either chemical precipitation (Loser Chemie) or electrolysis (Latunussa et al., 2016; Ardente et al., 2019). For example, Yang et al. (2017) introduced HCl to make AgCl precipitates (ca. 70% purity), which underwent subsequent electrochemical refinement to obtain 99.995% purity.

2.3.4. Wastewater treatment and landfilling

The last stage of PV recycling is usually wastewater treatment, incineration, and landfilling. Both hydrometallurgy and electrolysis processes produce large amounts of wastewater and sludge containing various metals. A further neutralization process is required when strong acids or bases are used (Latunussa et al., 2016; Ardente et al., 2019). Plastics are subjected to incineration, and the leachate, after recovering precious metals, is sent to a separate wastewater treatment process, of which the subsequent residual solids (fly/bottom ashes and metal sludge) are beneficially reused (e.g., clinker production) (Corcelli et al., 2018) or otherwise they need to be properly landfilled in the end.

Table 2.1. Various commercialized and proposed PV panel (cell) recycling processes with reclaimed resources ('-' means not mentioned).

Process	Delamination (low-rate recovery)		Recovery from solar cell (high-rate recovery)		
	Module separation	EVA removal	Leaching & etching	Selective recovery	Additional process (chemical used)
Recovered Materials	Metal frame, copper cable, junction box	Glass, cell, back sheet, heat	Mixed metal solution, Si wafer	Ag, Cu, Al	
FRELP[a] (SASIL)	Mechanical separation	Incineration	Hydrometallurgy (HNO_3)	Electrolysis	Neutralization ($Ca(OH)_2$) required
ELSi[b] (Fraunhofer)	Mechanical separation	Pyrolysis followed by solvent dissolution	Electrochemical processes	-	-
NEDO	Mechanical separation	Pyrolysis / Hot knife	-	-	-
SUPER PV (Loser Chemie)	Mechanical separation	Optical nanotechnology -	Hydrometallurgy (MSA; $H\text{-}OSO_2CH_3$)	Chemical precipitation	-
Corcelli et al. (2018)	Mechanical separation	Thermal treatment (20 ˚C to 600 ˚C; stepwise increase)	Hydrometallurgy (HCl)	-	Neutralization ($Na(OH)_2$) required
Yang et al. (2017)	-	-	Hydrometallurgy (MSA)	Chemical precipitation	Chemical & electro-refining (NaOH, H_2O_2)

[a]FRELP: Full recovery of end-of-life PV panel
[b]ELSi: Industrial scale recovery and reuse of all materials from end-of-life silicon-based photovoltaic modules
[c]NEDO: New energy industrial technology development organization
Ag: silver; Cu: copper, Al : aluminum

2.4. LCA approaches for the EoL c-Si PV panel

2.4.1. Past and recent trends in life cycle assessment studies for c-Si PV systems

In the 1990s, many baseline studies were conducted to set the basis of the LCA framework for PV system design and to evaluate the systems bottlenecks and potentials (Phylipsen and Alsema, 1995; Keoleian and Lewis, 1997; Aguado-Monsonet, 1998). Since the development of the PV system was at an early stage, the LCA scope was limited and only focused on elucidating the positive environmental impacts of c-Si technologies with respect to energy consumption and emissions during the manufacturing and installation of PV systems. According to Frankl et al. (1998), a simplified LCA approach of building-integrated PV systems was more favorable compared to conventional PV power plants with respect to energy generation and greenhouse gas emissions. Dones and Frischknecht (1998), who performed LCA studies of mono- and polycrystalline silicon PV module technologies using environmental inventories focused on material/energy requirements and emissions, reported similar results. They concluded that the electricity requirement during the manufacturing process was the most important contributor to the environmental burden. Their results provided useful information for future LCA research on energy requirements and GHG emissions in the PV manufacturing process. Nonetheless, both studies were restricted to manufacturing and energy production, but the environmental impacts of mining of raw materials, transportation, and recycling of PV panels were not considered.

In order to further clarify the environmental impact of PV systems, studies in the following decade expanded the boundaries of LCA analysis to consider other environmental impacts during the entire life cycle, i.e. ‘cradle-to-grave’ approach, including EoL scenarios (Fthenakis, 2004; Müller et al., 2005). For example, Jungbluth (2005) conducted an LCA study based on the Swiss life cycle inventories for PV power plants with process data including quartz reduction, silicon purification, wafer, panel, and laminate production and mounting structure with 30 years of operation. Müller et al. (2005) performed LCA analyses with the inclusion of a recycling process for c-Si PV modules based on the inputs obtained from the operation at Deutsche Solar. The results showed that the recycling of PV modules mitigated the environmental impacts compared to other EoL scenarios, such as incineration and disposal at a landfill. Furthermore, the economic views and the sensitivity analysis have been included in LCA by few researchers (Kannan et al., 2006; Pacca et al., 2007; Ito et al., 2008). Ito et al. (2008) studied the LCA of various PV modules such as multi-crystalline silicon (m-Si) and amorphous silicon (a-Si) to investigate their environmental impacts and cost-effectiveness. This study indicated that the PV module efficiency was a primary factor in improving energy production and preventing greenhouse gas emissions. In addition, the sensitivity analysis to ambient temperature revealed that the m-Si module performance ratio decreased by 10% as the temperature increased from 5.8 to 30.2 °C, which affected the energy payback time (EPBT) and CO_2 emission rate.

Recently, more studies have been designed to explore the impact of a proactive approach to improve the treatment of EoL PV panels. To examine the sustainability and environmental benefits of PV systems, EPBT and GHG emission rate were extensively adopted as environmental indicators. EPBT is defined as the time required to recover the energy invested in the system, which is one of the most adopted metrics to characterize the energy sustainability of PV system. GHG emissions over a given life cycle is an important metric, indicating its environmental impact on global warming. Since LCA has three main aspects; energy, environmental, and economic, EPBT and GHG emissions have been frequently estimated to determine the benefits in terms of energy and environment compared to conventional alternatives (Armendariz-Lopez et al., 2018; Yue et al., 2012).

The EPBT and GHG emissions were dependent on several key parameters such as PV technology, module-rated efficiency and degradation rate, irradiation of the location, and a lifetime of the PV. In this review, the authors demonstrated that among the five types of solar cells, mono-crystalline Si, multi-crystalline Si, amorphous Si, cadmium telluride thin film, and copper-indium-gallium-selenide thin film, the mono-crystalline Si PV systems exhibited the worst environmental performance due to high-energy consumption during the production processes (Peng et al., 2013). Sherwani et al. (2010) had previously reviewed the LCA of EPBT and GHG emissions of solar PV systems while considering mass and energy flow from silica extraction to panel assembly. Both review papers remarked that increasing solar cell efficiency and material recycling rates would further improve the environmental performance of PV systems by reducing the energy requirement and GHG emissions. In line with this view, a few researchers have studied the environmental impacts of PV systems that are coupled with recycling processes.

Recent research trends associated with evaluating and comparing environmental impacts using LCA methods for silicon PV technologies focused on material recycling are discussed in more detail later in this section.

2.4.2. Environmental impacts of EoL c-Si PV panel based on LCA perspective

According to the IRENA report (2016), the total volume of PV waste in circulation will be approximately 1.7 to 8.0 million tons by 2030, and it is estimated to reach 60–78 million tons by 2050. As mentioned above, the first-generation c-Si modules have occupied more than 80% of the market share worldwide over the last 30 years, and thus contribute to a substantial portion of the PV waste stream. Considering the increase in PV installation and waste generation, recent studies on c-Si PV waste treatment and recycling processes assessed with LCA methodology for environmental impacts are introduced in the following section. Table 2.2 summarizes the currently available literature that applied LCA to the EoL phase of c-Si PV panels by comparing functional units, system boundaries, software/databases used, impact analysis methodologies, and main contents. It is always difficult to directly compare the life cycle impact analysis (LCIA) results from different studies because of the differences in the functional unit, system boundary, and LCIA methodology. In addition, the degree of "recycle" varies for different studies. For example, Vellini et al. (2017) only considered the recovery of Si (90% efficiency) to make new wafers, whereas Corcelli (2018) included glass, Al frame, heat, and precious metal recoveries from the cell surface in the contents of recycling. Bogacka et al. (2017), Duflou et al. (2018), Dias et al. (2021) conducted their analysis up to the endpoint by simplifying multiple midpoint indicators into a few endpoint indicators (e.g., human health, ecosystems, and resources depletion), considering complex cause-effect chain, while the remaining six applied various types of mid-point analysis with respect to each of the LCIA methods used (ReCiPe, CML, and ILCD). Depending on the system boundary, the existing literature is divided into three groups: (1) Four out of ten have a system boundary from production to EoL, which regards recycling as a unit process being compared with incineration or landfill; (2) six out of ten focused only on recycling having a series of sequential sub-processes, and (3) six studies interpreted the LCIA results in disaggregated form so that the relative contribution of each sub-process to entire recycling could be individually evaluated, and four studies considered collection of PV waste and the effect of transport to the recycling facility.

Table 2.2. Comparison of current literature that applied life cycle assessment (LCA) to the end-of-life (EoL) phase of c-Si photovoltaic (PV) panels.

Recycling Process (scale)	**Functional Unit**	**System boundary**	**Software /DB**	**LCIA[a] methodology (mid/endpoint)**	**Comparative objective (recovered materials)**	**LCA results by process**	**Authors**
Deutsche Solar (pilot scale)	72 cells (125 mm × 125 mm)	EoL (excluded collection)	SimaPro/ Ecoinvent 2000	CML Baseline-2000 (midpoint)	Incineration vs. recycling (Si wafer)	Aggregated	Müller et al. (2005)
Conventional process (literature-based)	1 m^2	Production to EoL (excluded collection)	GaBi/ Ecoinvent 2.2	CML 2001, EPBT, (midpoint)	With vs. W/O recycling (Si wafer)	Aggregated	Vellini et al. (2017)
Literature based (literature-based)	1 kW	Production to EoL (excluded collection)	GaBi	ReCiPe, (midpoint)	Landfill vs. recycling (Al, Ag, glass and Si)	Aggregated	Huang et al. (2017)
High rate vs. low rate recovery process (lab scale)	1 m^2	EoL (excluded collection)	SimaPro/ Ecoinvent 3.1	ReCiPe, (midpoint)	2 recycling scenarios (frame, glass, heat, Al, Si and Cu)	Partially disaggregated	Corcelli et al. (2018)

Destructive separation, thermal, and chemical treatment, and selective delamination (literature-based)	1 kg of PV waste	EoL (excluded collection)	SimaPro/ Ecoinvent 3.3	ReCiPe, (both mid and endpoint)	3 recycling scenarios (frame, glass, Cu and Ag)	Partially disaggregated	Duflou et al. (2018)
Conventional process (literature-based)	0.65 m^2	Production to EoL (excluded collection)	SimaPro/ Ecoinvent 3.0	ReCiPe (both mid and endpoint)	n/a	Aggregated	Bogacka et al. (2017)
FRELP process (pilot scale)	1,000 kg of PV waste	EoL (included collection)	SimaPro / Ecoinvent 3.0	ILCD (15 midpoint categories)	Relative contribution on each sub-processes	Fully disaggregated	Latunussa et al. (2016)
			SimaPro/ Ecoinvent 2.0	ILCD (16 midpoint categories)	Central vs. Decentralization of recycling facility (frame, Al, Cu, heat, glass, Si and Ag)	Partially disaggregated	Ardente et al. (2019)

Thermal, mechanical, and chemical delamination, (literature-based)	1 m^2	Production to EoL (excluded collection)	GaBi 8.1/ Ecoinvent 3.3	TRACI (midpoint)	Six different delamination methods (glass, Si, Al)	Partially disaggregated	Maani et al. (2020)
Organic solvent-thermal-chemical treatment (lab scale)	5W c-Si module	EoL (excluded collection)	OpenLCA/Ecoinvent3.2	ReCiPe/ (endpoint)	Experimental recovery vs. landfilling (glass, Al, Si, Ag, Cu)	Fully disaggregated	Dias et al., (2021)

[a]LCIA: Life cycle impact analysis
Al: aluminum, Si: silicon, Ag: silver, Cu: copper

2.4.2.1. From production to EoL

Based on two different scenarios, disposal and recycling, the energy and environmental analysis of c-Si panels throughout the entire PV panel production process have been compared by Vellini et al. (2017). The recycling process was divided into two steps: (1) thermal process for the separation of EVA-laminated cells, and (2) chemical process for recycling of silicon and sheets. Compared with the disposal scenario, the recycling of the c-Si panel reduced the terrestrial eco-toxicity potential and GWP by 73.6% and 24.0%, respectively. During the disposal step, the contribution to human and terrestrial toxicity has been mainly driven by lead and copper emissions to soil, respectively. Thus, a PV panel recycling process drastically reduces the two impact categories as compared to landfilling. The reduction in the GWP of the recycling scenario was mainly attributed to lower energy consumption during the purification phase, while the recycling step required high quantities of energy and heat. Additionally, the entire stages of a panel recycling process have contributed to the minimization of other environmental burdens.

Huang et al. (2017) introduced the environmental impact assessment of the recycling process of EoL c-Si PV panel in China. This study compared the environmental impacts between landfill and recycling scenarios throughout the entire life cycle of the PV module from manufacturing to EoL (i.e., cradle-to-grave). The results of environmental impact comparison demonstrated that the environmental burdens of the recycling process were much less than those of the landfill scenario. This was mainly linked to the material reduction resulting from the recycling process. Even though the recycling process consisted of a dismantling step, thermal, and chemical treatment, the environmental burdens were still lower than in the landfill scenario. These findings are in line with those of other studies (Müller et al., 2005; Vellini et al., 2017). Regarding a comparatively lower recycling rate in China, they concluded that further efforts would be made to improve the recycling technology.

2.4.2.2. Comparison of EoL under different scenarios

Müller et al. (2005) primarily and specifically performed the LCA study with consideration of EoL phase for c-Si modules using the data obtained from a pilot-scale operation of “Deutsche Solar” recycling process. In this study, the recycling process consisted of thermal and chemical processes. The recycling process began with burning off the laminate followed by chemical etching for metallization and separation. According to their LCA results, the avoidance of a new wafer production mostly contributed to the environmental disburdens. It was also reported that the environmental burden mainly resulted from the energy consumption during the thermal treatment and the use of chemicals in the etching line. Compared with an incineration-landfill scenario, the environmental disburden of the incineration scenario was ascribed only to recycling the aluminum frame, while there was no environmentally positive impact from the reuse of wafers and glass. Compared to a shredder process followed by sorting of materials such as glass, plastics, and metals, their recycling process, even with higher energy consumption, showed a positive response due to the recycling benefits.

Duflou et al. (2018) performed a comparative study based on three different scenarios: baseline (destructive separation), thermal and chemical treatment, and delamination (selective mechanical separation) scenarios. Selective mechanical delamination through milling and cleaving illustrated a substantial reduction in the environmental impact caused by improvement in resource recovery. According to the environmental impact analysis of the delamination scenario, evading metal depletion became most apparent and effective among the several categories, which resulted from maximizing the recovery of silver and copper. Compared to the thermal and chemical treatment scenarios, the absence of pyrolysis in the delamination scenario played a major role in further reducing the environmental impacts related to material substitution.

2.4.2.3. Comprehensive approaches for the recycling process

The sustainability of a recovery process for EoL c-Si PV panels has been comprehensively evaluated by Corcelli et al. (2018) using the LCA indicators associated with material recovery and energy savings. In this study, the system boundary contained two subsystems: (1) thermal treatment of the decommissioned PV panel, and (2) recycling of recoverable fractions under two different scenarios: high- and low-rate recovery. The main difference between these two scenarios was the recycling of materials such as silicon and copper, which were subjected to the production of secondary raw materials (SRMs). In this regard, their system has been expanded to recognize the recycling of materials for secondary production and the resulting energy savings. Corcelli et al. (2018) made a detailed analysis of the contribution of each single stage for the entire recycling process to the environmental impact. Interestingly, their results showed that the environmental benefits under the high-rate recovery scenario were mostly driven by the recovery of aluminum and silicon, resulting from refining the recovered products to harvest SRMs. In particular, the environmental disburden associated with freshwater eutrophication, human toxicity, terrestrial acidification, and fossil depletion were shown to be strongly pronounced among all impact categories. In order to improve the sustainability of the life cycle of PV systems, they have recommended a well-established recycling process responsible for the efficient recovery of precious materials such as silicon and silver.

Latunussa et al. (2016) created an onsite dataset (i.e., life cycle inventories) based on an actual pilot scale (1 t/h up to 8,000 t/yr.) PV panel-recycling project, “Full Recovery of End-of-Life Photovoltaic project (FRELP).” The authors applied LCA to evaluate the environmental impact of recycling 1,000 kg of PV waste (i.e., functional unit) at each sub-stage from transportation to disposal of residues, with the consideration of internal cable as the balance of system. Energy recovered from thermal treatment (i.e., incineration) is reflected as a negative value (i.e., avoided impact). Results are presented in a disaggregated data, unlike most previous studies, which shows the contribution of each sub-stage to the entire recycling process. Transport of waste PV, incineration of plastics, and metal recovery process (i.e., sieving of bottom ash-leaching-electrolysis-neutralization) occupied a noticeable portion of the total environmental impact. In the case of climate change as a representative impact category, incineration of the PV sandwich made the largest contribution (34%), followed by transport of the PV waste (29%) and metal recovery process (24%).

Furthermore, advances were made by Ardente et al. (2019) where additional consideration is given to the environmental credits from the recovered SRMs, including precious metals such as Ag, Si, and Cu. Compared to conventional recycling (base case, only Al frame is recycled), the avoided impact from the recovery of the raw material counterbalances the impacts from the high-rate recycling process, especially from the resource depletion category. The authors also emphasized the impact of transport of PV waste by comparing the centralization and decentralization of recycling facilities, concluding that the primary separation of glass, frame, and cables needs to be accomplished in a decentralized facility located near the collection point, which can reduce approximately 80% of the total weight of PV waste and subsequent impacts from all categories (more than 10%). Similarly, Goe and Gaustad (2016) also presented a combined LCA and geospatial modeling to minimize the environmental burden from both transport and recycling of EoL PV panels, which depends on the size of cities.

Comparative studies on the specific component technologies were also carried out. Maani et al. (2020) conducted a comparative LCA on the six different delamination methods (i.e., nitric acid dissolution, solvent and ultrasonic irradiation, solvent dissolution, thermal treatment, electro thermal heating, and pyrolysis) comparing extraction of virgin metal, indicating that the currently available recycling technologies are not always environmentally sustainable (e.g., nitric acids leaching showed the highest impacts). Dias et al., (2021) conducted a combined economic-environmental analysis on the recycling process comprised of organic solvent (i.e., toluene) delamination followed by thermal decomposition and nitric acid leaching. The results show that the recycling of Al frame and junction box is profitable, whereas the downstream metal recovery process is unlikely to be profitable due to the labor costs despite the positive environmental impacts, which is consistent with Faircloth et al. (2019) indicating that PV waste flow rate, and initial investment are the major affecting factors in the case of Thailand. Franz and Piringer (2020) also conducted combined environmental and economic analysis on the EoL of PV systems concluding that greater attention needs to be paid on the release of metals from broken PV module, and decommissioning of free-field PV systems.

Currently, a meta-analysis called "harmonization" has been proposed to effectively reduce the variability and help clarify the central tendency of the various estimates on different PV systems (Kim et al., 2020). Also, recently published guidelines for the recycling (i.e., WEEE directive) (Wambach et al., 2017) and LCA of c-Si PV panel (IEA, 2020) provides consistency and rationales for the localized PV-specific parameters that can be used in life cycle inventory However, the EoL phase and a universal standard across various LCA studies to minimize variability are not yet available. For such a reason, only three studies from Table 2.2 (Müller et al., 2005; Corcelli et al., 2018; Ardente et al., 2019) were available for the comparison; using conversion factors (e.g. 22 kg of PV waste is equivalent to 1.6 m^2 of panel surface from Latunussa et al. (2016)) provided in the paper, normalized effective GHG emission (i.e., burden from the recycling processes – disburden from the avoided impacts) based on the equivalent functional unit of 1 m^2 PV panel was found to be -52.62, -60.5, and - 26.61 kg CO_2 eq respectively. The substantial difference between the first two and the last value is presumably due to the differences in system boundary; that is, the collection of waste panel (transportation effect) is included Ardente et al. (2019) whereas the first two studies excluded it from their system boundary.

This chapter has been published as a review paper in Waste Management (128, 45-54), 2021.

References

Aguado-Monsonet, M.A., 1998. The environmental impact of photovoltaic technology. Sevilla: Institute for Prospective Technological Studies 7-8.

Ardente, F., Latunussa, C.E.L., Blengini, G.A., 2019. Resource efficient recovery of critical and precious metals from waste silicon PV panel recycling. Waste Manage. 91, 156-167.

Armendariz-Lopez, J. F., Arena-Granados, A. P., Gonzalez-Trevizo, M. E., Luna-Leon, A., & Bojorquez-Morales, G., 2018. Energy payback time and Greenhouse Gas emissions: Studying the international energy agency guidelines architecture. J. Clean. Prod. 196, 1566–1575.

Bogacka, M., Pikon´, K., Landrat, M., 2017. Environmental impact of PV cell waste scenario. Waste Manage. 70, 198-203.

Bracquene, E., Peeters, J.R., Dewulf, W., Duflou, J.R., 2018. Taking evolution into account in a parametric LCA model for PV Panels. Proc. CIRP 69, 389-394.

Chung, J.B., Kim, E.S., 2018. Public perception of energy transition in Korea: nuclear power, climate change and party preference. Energy Policy 116, 137-144.

Chung, J., Seo, B., Lee, J., Kim, J.Y., 2021. Comparative analysis of I2-KI and HNO3 leaching in a life cycle perspective: towards sustainable recycling of end-of-life c-Si PV panel. J. Hazard. Mater. 404, 123989.

Corcelli, F., Ripa, M., Leccisi, E., Cigolotti, V., Fiandra, V., Graditi, G., Sannino, L., Tammaro, M., Ulgiati, S., 2018. Sustainable urban electricity supply chain - indicators of material recovery and energy savings from crystalline silicon photovoltaic panels end-of-life. Ecol. Indic. 94, 37-51.

Cossu, R., Williams, I.D., 2015. Urban mining: concepts, terminology, challenges. Waste Manage. 45, 1–3.

Dale, M., Benson, S.M., 2013. Energy balance of the global photovoltaic (PV) industry - Is the PV Industry a Net Electricity Producer? Environ. Sci. Technol. 47, 3482-3489.

Deng, R., Chang, N.L., Ouyang, Z., Chong, C.M., 2019. A techno-economic review of silicon photovoltaic module recycling. Renew. Sustain. Energy Rev. 109, 532-550.

Devabhaktuni, V., Alam, M., Depuru, S.S.S.R., Green II, R.C., Nims, D., Near, C., 2013. Solar energy: Trends and enabling technologies. Renew. Sustain. Energy Rev. 19, 555-564.

Dias, P., Schmidt, L., Lunardi, M.M., Chang N, L., Spier, G., Corkish, R., Viet, H., 2021 Comprehensive recycling of silicon photovoltaic modules incorporating organic solvent delamination – technical, environmental and economic analyses. Resour. Conserv. Recycl. 165, 105241.

Dones, R., Frischknecht, R., 1998. Life-cycle assessment of photovoltaic systems: results of Swiss studies on energy chains. Prog. Photovoltaics 6, 117-125.

Duflou, J.R., Peeters, J.R., Altamirano, D., Bracquene, E., Dewulf, W. 2018. Demanufacturing photovoltaic panels: Comparison of end-of-life treatment strategies for improved resource recovery. CIRP Annals 67, 29-32.

European Commission (EC), 2012. Directive 2012/19/EU of the European Parliament and of the Council of 4 July 2012 on waste electrical and electronic equipment (WEEE).

Flowers M.E., Smith M.K., Parsekian A.W., Boyuk D.S., McGrath J.K., Yates L., 2016. Climate impacts on the cost of solar energy. Energy Policy 94, 264-273.

Frankl, P., Masini, A., Gamberale, M., Toccaceli, D., 1998. Simplified life-cycle analysis of PV systems in buildings: Present situation and future trends. Prog. Photovoltaics 6, 137-146.

Franz, M., Piringer, G., 2020. Market development and consequences on end-of-life management of photovoltaic implementation in Europe. Energy Sustain. Soc. 10, 1−-21.

Fraunhofer ISE, 2020. Photovoltaics report, freiburg. Available at: ttps://www.ise. fraunhofer.de/content/dam/ise/de/documents/publications/studies/Photovoltaics-Report.pdf. (Accessed 7 June 2019).

Fthenakis, V.M., 2004. Life cycle impact analysis of cadmium in CdTe PV production. Renew. Sustain. Energy Rev. 8, 303-334.

Goe, M. and G. Gaustad, 2016. Estimating direct climate impacts of end-of-life solar photovoltaic recovery. Sol. Energy Mater. Sol. Cells 156, 27-36.

Hauschild, M.Z., Huijbregts, M.A.J. (Eds.), 2015. Life cycle impact assessment.

Springer, Netherlands, Dordrecht.

Huang, B., Zhao, J., Chai, J., Xue, B., Zhao, F., Wang, X., 2017. Environmental influence assessment of China's multi-crystalline silicon (multi-Si) photovoltaic modules considering recycling process. J. Sol. Energy 143, 132-141.

International Energy Agency (IEA), 2009. Methodology guidelines on Life Cycle Assessment of photovoltaic electricity.

International Energy Agency (IEA), 2019. World Energy Outlook 2019, IEA, Paris.

International Energy Agency (IEA), 2020. Snapshot of global PV markets.

International Renewable Energy Agency (IRENA), 2016. End-Of-Life management: solar photovoltaic panels.

International Renewable Energy Agency (IRENA), 2019. Global energy transformation: A roadmap to 2050 (2019 edition), International Renewable Energy Agency, Abu Dhabi.

International Renewable Energy Agency (IRENA), 2020. Renewable capacity statistics 2020.

International Organization for Standardization (ISO), Standard ISO 14040 Environmental management – Life cycle assessment – Principles and frame- work, 2006.

Ito, M., Kato, K., Komoto, K., Kichimi, T., Kurokawa, K., 2008. A comparative study on cost and life-cycle analysis for 100 MW very large-scale PV (VLS-PV) systems in deserts using m-Si, a-Si, CdTe, and CIS modules. Prog. Photovoltaics 16, 17-30.

Jordan, D.C., Kurtz, S.R., 2013. Photovoltaic degradation rates - an analytical review. Prog. Photovoltaics 21, 12-29.

Jungbluth, N., 2005. Life cycle assessment of crystalline photovoltaics in the Swiss ecoinvent database. Prog. Photovoltaics 13, 429-446.

Kannan, R., Leong, K.C., Osman, R., Ho, H.K., Tso, C.P., 2006. Life cycle assessment study of solar PV systems: An example of a 2.7 kWp distributed solar PV system in Singapore. J. Sol. Energy 80, 555-563.

Keoleian, G.A., Lewis, G.M., 1997. Application of life-cycle energy analysis to photovoltaic module design. Prog. Photovoltaics 5, 287-300.

Korea Institute for Industrial Economics and Trade (KIET), 2017. Revitalization of renewable energy industry through post-management system development

Korea Institute of Energy Research (KIER), 2015. A research on photovoltaic module recycling system development, 118-144.

Korea Environment Institute (KEI), 2018. Management status and improvement plans of waste solar panels

Kuczyńska-Łażewska, A., Klugmann-Radziemska, E., Sobczak, Z., Klimczuk, T., 2018 Recovery of silver metallization from damaged silicon cells. Sol. Energy Mater. Sol. Cells 176, 190-195.

Latunussa, C.E.L., Ardente, F., Blengini, G.A., Mancini, L., 2016. Life Cycle Assessment of an innovative recycling process for crystalline silicon photovoltaic panels. Sol. Energy Mater. Sol. Cells 156, 101-111.

Lewis, N.S., 2007. Toward cost-effective solar energy use. Science 315, 798-801.

Liu, L., Jia, Y., Lin, Y., Alva, G., Fang, G., 2017. Performance evaluation of a novel solar photovoltaic–thermal collector with dual channel using microencapsulated phase change slurry as cooling fluid. Energy Convers. Manage. 145, 30-40.

Ministry of Environment (ME), 2019. Act on resource circulation of electrical and electronic equipment and vehicles. Republic of Korea.

Ministry of Trade, industry, and energy (MOTIE), 2017. Renewable energy 3020 implementation plan. Republic of Korea.

Müller, A., Wambach, K., Alsema, E., 2005. Life cycle analysis of solar module recycling process. In: Proceedings of the 20th European Photovoltaic Solar Energy Conference, Barcelona, Spain, 6–10 June 2005.

Ozoemena, M., Hasan, R., Cheung, W.M., 2016. Analysis of technology improvement opportunities for a 1.5 MW wind turbine using a hybrid stochastic approach in life cycle assessment. Renew. Energy 93, 369-382.

Pacca, S., Sivaraman, D., Keoleian, G.A., 2007. Parameters affecting the life cycle performance of PV technologies and systems. Energy Policy 35, 3316-3326.

Paiano, A., 2015. Photovoltaic waste assessment in Italy. Renew. Sustain. Energy

Rev. 41, 99-112.

Palitzsch, W., Schonherr, P., Loser, U., 2014. Integrated approach for economic PV waste recycling. 2014 IEEE 40th Photovolt. Spec. Conf. PVSC 2014, 2709-2711.

Peeters, J.R., Altamirano, D., Dewulf, W., Duflou, J.R. 2017. Forecasting the composition of emerging waste streams with sensitivity analysis: A case study for photovoltaic (PV) panels in Flanders. Resour. Conserv. Recycl. 120, 14-26.

Peng, J., Lu, L., Yang, H., 2013. Review on life cycle assessment of energy payback and greenhouse gas emission of solar photovoltaic systems. Renew. Sustain. Energy Rev. 19, 255-274.

Perpiñan, O., Lorenzo, E., Castro, M.A., Eyras, R., 2009. Energy payback time of grid connected PV systems: comparison between tracking and fixed systems. Prog. Photovoltaics 17, 137-147.

Phylipsen, G.J.M., Alsema, E.A., 1995. Environmental life-cycle assessment of multicrystalline silicon solar cell modules, Department of Science, Technology, and Society, Utrecht University Utrecht.

Ross, R.G., 2014. PV reliability development lessons from JPL's flat plate solar array project. IEEE J. Photovolt. 4, 291-298.

Santos, J. D., & Alonso-García, M. C., 2018. Projection of the photovoltaic waste in Spain until 2050. Journal of Cleaner Production, 196.

Sauser, W. I. Jr., L., & Sims, R., 2014. Ethical Issues in Electronic Waste Disposal:

Philosophical Analysis and Proposed Solutions. Journal of Management, 15(2), 11–25.

Sherwani, A.F., Usmani, J.A., Varun, 2010. Life cycle assessment of solar PV based electricity generation systems: A review. Renew. Sustain. Energy Rev. 14, 540-544.

Shi, X., Qian, Y., Yang, S., 2020. Fluctuation analysis of a complementary wind-solar energy system and integration for large scale hydrogen production, ACS Sustainable Chem. Eng. 8, 7097-7110.

Sica, D., Malandrino, O., Supino, S., Testa, M., & Lucchetti, M. C., 2018. Management of end-of-life photovoltaic panels as a step towards a circular economy. Renewable and Sustainable Energy Reviews, 82(December 2016), 2934–2945.

SolarPower Europe, 2018. Global market outlook for solar power 2018-2022. Retrieved.

Tyagi, V.V., Rahim N.A.A., Rahim, N.A., Jeyraj, A., Selvaraj, L., 2013. Progress in solar PV technology: research and achievement. Renew. Sustain. Energy Rev. 20, 443-461.

Vellini, M., Gambini, M., Prattella, V., 2017. Environmental impacts of PV technology throughout the life cycle: Importance of the end-of-life management for Si-panels and CdTe-panels. Energy 138, 1099-1111.

Wambach, K., Heath, G. & Libby, C., 2017. Life Cycle Inventory of Current Photovoltaic Module Recycling Processes in Europe IEA-PVPS Task 12 Report #T12–T12:2017 (International Energy Agency Photovoltaic Power Systems

Programme, 2017).

Weitemeyer, S., Kleinhans, D., Vogt, T., Agert, C., 2015. Integration of Renewable Energy Sources in future power systems: the role of storage. Renew. Energy 75, 14-20.

Yang, E.H., Lee, J.K., Lee, J.S., Ahn, Y.S., Kang, G.H., Cho, C.H., 2017. Environmentally friendly recovery of Ag from end-of-life c-Si solar cell using organic acid and its electrochemical purification. Hydrometallurgy 167, 129-133.

Yue, D., Khatav, P., You, F., Darling, S.B. 2012. Deciphering the uncertainties in life cycle energy and environmental analysis of organic photovoltaics. Energy Environ. Sci. 5(11), 9163–9172.

Chapter 3. End-of-life scenarios of c-Si PV panels considering environmental impacts: comparative analysis of high- and low-level scenarios

3.1. Introduction

The total amount of end-of-life (EoL) photovoltaic (PV) panels will exponentially grow from 0.1 million tons in 2016 to 60–78 million tons in 2050 (IRENA, 2016). In 2012, the European waste electrical and electronic equipment directive (WEEE 2012/19/EU) had added PV panels to the list of electric and electronic equipment, requiring the adequate treatment of EoL PV panels (EC, 2012). Recently, the Korean government amended the Act on the Resource Circulation of Electrical and Electronic Equipment and Vehicles, which was expanded to include solar PV panels as one of the 23 extended producer responsibility (EPR) items (ME, 2019). This amendment requires solar panel manufacturers and importers to be responsible for their recycling. However, implementing the proposed amendment has been postponed to 2023 because of a lack of the necessary recycling infrastructure.

For the next few decades, crystalline-silicon (c-Si) PV technology will dominate the market (Fraunhofer ISE, 2020). Thus, previous studies have proposed reuse, recycling, incineration, and landfill as options for an adequate PV module EoL approach (Lunardi et al., 2018, Held, 2013). EoL PV panel landfills can generate noticeable pollution via toxic metal leaching into the environment, such as lead and silver. Incinerating PV panels can impose severe environmental burdens, such as releasing toxic heavy metals into the atmosphere (Müller et al., 2005). Reusing PV modules by replacing them with a new aluminum frame and junction box would decrease their efficacy as time passes. This approach would only delay the environmental impacts associated with EoL PV panels rather than solve the problem (Lunardi et al., 2018).

Recycling EoL PV panels starts with disassembling the aluminum frame and junction box. The next steps include removing ethylene-vinyl acetate (EVA) encapsulants between the glass and PV cells for glass separation (delamination) and separating materials from PV cells. Until now, many studies have been conducted to develop economic and environmentally friendly recycling technologies, including comparison studies to assess their environmental burdens and cost-effectiveness (Deng et al., 2019, Maani et al., 2020). Mechanical, chemical, and thermal methods have been proposed for delamination, and the materials are manually and chemically separated from PV cells.

Sufficient life cycle inventory data and information associated with the recycling process should be cataloged to achieve sustainable EoL PV panel recycling. The environmental impact would be assessed to find the most cost-effective and environmentally friendly recycling method. The recycling process for recovering materials is critical for sustaining the growing number of EoL PV panels. Only a few studies have focused on applying LCA to analyze the potential benefits and impacts of recovering glass and metals, such as aluminum and copper (Held, 2013). Peeters et al. (2017) have highlighted the importance of reliable prediction, such as EoL PV panel volume and composition. PV cell efficacy increases with the date of installation as substances contained in the PV cells improve. Thus, a few studies have been conducted to estimate the PV waste volume considering the changes in their composition (Santos and Alonso-García, 2018; Peeters et al., 2017; Paiano, 2015).

Detailed data and an inventory for the recycling process of PV modules have been introduced in Europe recently (Wambach, 2018; Stolz and Frishcknecht, 2018). However, only a few life cycle inventories (LCI) are available for PV module recycling at an industrial scale. Even though LCI for PV module production has been thoroughly established and updated, the database for assessing EoL PV recycling is limited. Thus, the environmental burdens of each recycling process can be accurately evaluated using these limited datasets. Some reports did not even indicate the input and output data of each step in the recycling process. Consequently, their environmental impacts were aggregated to the entire recycling process only.

In Korea, the Ministry of Commerce, Industry, and Energy plans to complete the nation’s first PV module recycling center in Jincheon, Chungcheongbuk-do, in September 2021 to treat 3600 tons of waste yearly. Because this recycling center is still under construction, it is currently impossible to evaluate it using LCI analyses. However, it is necessary to provide recycling LCI and evaluate environmental impacts for heat treatment to be used and applicable material recovery technologies.

In this chapter, therefore, EoL scenarios are first established for applicable processes in Korea, and their environmental impacts are compared by using LCA methodology.

Particularly, the environmental impacts of different EoL treatment options are compared and analyzed at each process step with a comparison to conventional methods.

3.2 Methodology

In this study, the proposed scenarios were set over EoL of c-Si PV waste, including transportation, recycling, and final disposal, to investigate the environmental impact of the entire process using LCA. LCA, developed by the International Organization for Standardization (ISO) in the 1990s, quantifies all environmental impacts caused during a product or process's lifecycle (ISO, 2006). LCA includes four steps, namely, (1) goal and scope definition, (2) inventory analysis, (3) impact assessment, and (4) interpretation (Fig. 3.1). In this chapter, the LCA methodology was applied to the EoL phase of c-Si PV panels appropriate in Korea.

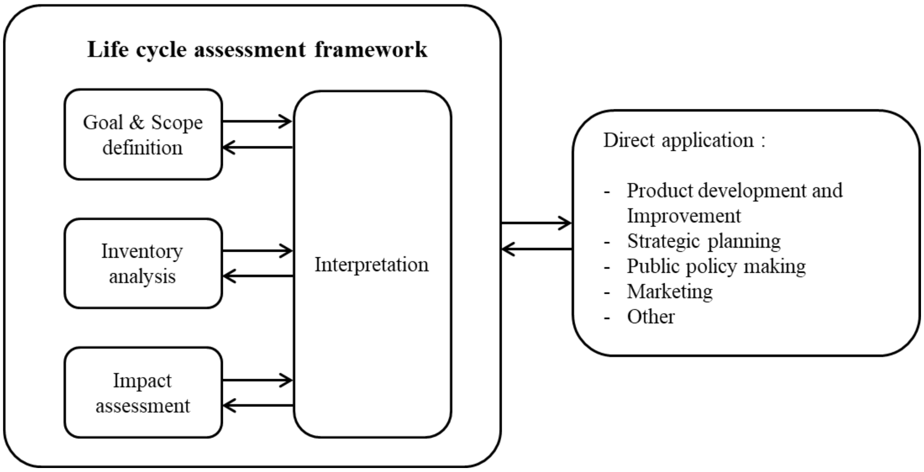


Fig. 3.1. Life cycle assessment framework (Source: ISO 14040, Environmental management–Life Cycle assessment–Principles and framework).

3.2.1. Material composition of c-Si PV waste

In general, the c-Si PV panel consists of the following structure, as shown in Fig. 3.1. Glass is used as the top layer of the c-Si PV panel to protect the PV cells from external damage and to allow the transmission of sunlight into the panel. It accounts for the highest mass percentage in the panel. The aluminum frame makes up the second largest percentage of mass. The role of the frames is to protect the panel and the glass layer. An EVA encapsulation layer is used to provide adhesive between the glass and the PV cells. It protects the PV cells from dirt, moisture, and other external shocks. The PV cell layer is composed of the light absorber (silicon) and electrical connections (silver, copper, lead, and other metals). In particular, the copper and lead in the composition of the c-Si PV waste are hazardous; thus, the PV waste containing these components is classified as waste that should be controlled properly according to the Basel Convention and should be subject to transboundary movement.

Polyvinyl fluoride (PVF) typically is used as a back-sheet layer that protects the PV cells from ultra-violet radiation, temperature changes, dirt, and moisture. Finally, the junction box is attached to the backside of the panel that carries the direct current produced from the cell to the inverter, which converts the direct current into alternating current (Maani et al., 2020; Duflou et al., 2018).

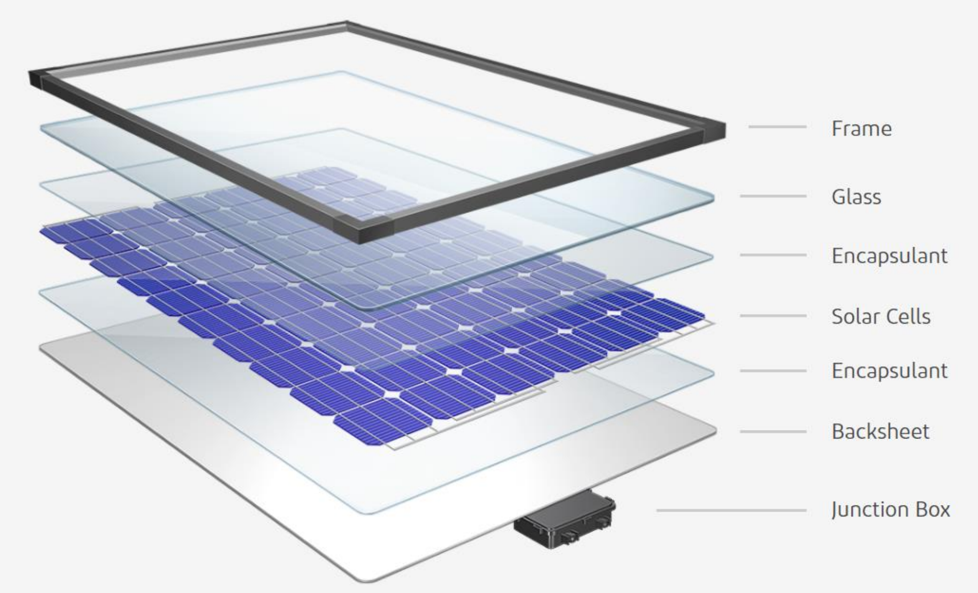


Fig. 3.2. Structure of a c-Si Panel (Source: htps://www.dupont.com/ products/what-makes-up-a-solar-panel.html).

The material composition of c-Si PV waste is different because the PV panel's composition varies with manufacturers and technological changes (Peeters et al., 2017). Maani et al. (2020) estimated the expected PV waste mass composition by averaging data from variable data sources. Most literature has not listed all PV panel compositions. IEA-PVPS T12 (Frischknecht et al., 2020) provides the bill of materials in detail. Table 3.1 lists the mass composition of the c-Si PV panel per ton used in this study.

Table 3.1. Material composition of the c-Si PV panel used in this study (Frischknecht et al., 2020; Latunussa et al., 2016; Maani et al., 2020; Stolz et al., 2016; Dias et al., 2021; Duflou et al., 2018; Wambach, 2018).

Component	Percentage (%, wt)	Percentage range(%, wt)
Glass	66.12	59.51 – 75.00
Al frame	15.95	7.82 – 20.00
EVA layer	6.56	4.50 – 7.00
Silicon	5.55	1.82 – 6.27
Back-sheet layer (PET 75.5%, PVF 24.5%)	3.60	0.80 – 3.77
Al	0.32	0.0 – 2.01
Cu	0.77	0.11 – 1.99
Ag	0.03	0.006 – 0.12
Other metals(Sn and Pb)	0.11	< 0.11
Cables (Cu 33%, polymer 67%)	1.00	0.75 – 4.75
Total	100	

Al: aluminum; EVA: ethylene-vinyl acetate; PET: polyethylene terephthalate; PVF: polyvinyl fluoride; Cu: copper; Ag: silver; Sn: tin

3.2.2. Goal and scope

The goal of this LCA was to assess the potential environmental impacts related to three different EoL treatment processes of c-Si PV waste to compare the processes and to identify the importance of various environmental factors. The functional unit (FU) of the LCA was 1 ton panels of c-Si PV waste with the mass composition depicted in Table 3.1.

The system boundary in this chapter is designed for three scenarios. Scenario 1 is a high-level EoL treatment, including thermal treatment for delamination and chemical treatment for material recovery. Scenario 2 is a low-level EoL treatment with final disposal in landfills, whereas Scenario 3 is the same but with final disposal in incineration plants. The next chapter explains the details of each EoL treatment process.

3.2.3. Description of c-Si PV EoL processes

As mentioned in Chapter 2, since the 1990s, attempts have been made to evaluate the environmental impacts through the entire life cycle of the PV system. According to the IEA guideline (IEA, 2020), the entire life cycle has been divided into product, construction, utilization, and EoL stages. And, the EoL stage includes dismantling, transport, waste processing, recycling and reuse, and disposal. In this study, the research boundary covers from the collection of PV panels that have reached the end of their lifespan to the final disposal.

The first step of EoL treatment of PV waste is transportation to the collection center. The next step includes the physical dismantling of the aluminum frame, cable, and junction box from the PV panels. This has been conducted either manually or automatically before initiating the main recycling process. At times, this work first can be done at a pre-treatment site (Wambach et al., 2018). The dismantled aluminum frame and copper wire can either be reused directly or, depending on their condition, sent to raw material producing facilities. And, the plastics separated from the cable and junction boxes are subjected to a subsequent final disposal process (i.e., incineration).

Under the European waste electrical and electronic equipment directive (WEEE

2012/19/EU), European recyclers already have been equipped with PV recycling facilities. European commercial recyclers mechanically treat c-Si PV modules to recover glass scrap after the aluminum frame and junction box are dismantled. The life cycle inventory data for these processes were presented in the study of Wambach et al. (2018), and the life cycle assessment was performed by Stolz and Frischknecht (2018).

Previously, high-efficiency recycling processes have been proposed, and, in addition to glass, aluminum, and copper, they can recover valuable materials within the PV cells (i.e., silicon, silver, copper, and aluminum) (Latunussa et al., 2016; Duflou et al., 2017; Klugmann-Radziemska et al., 2010; Wang et al., 2012). After disassembling the aluminum frame and junction box, the delamination step removes the EVA layer to separate the glass from the PV cells by thermal treatment, solvent dissolution, hot knife methods, and other methods.

After separating glass by delamination, silicon, silver, copper, and aluminum are recovered using a chemical treatment method. As mentioned in Chapter 2, the recovery of valuable materials was mainly performed using nitric acid (Latunussa et al., 2016; Huang et al., 2017, Maani et al., 2020) and alternatively via leaching agents such as methane sulfonic acid (MSA) and iodine-iodide solutions (Palitzsch et al., 2014; Yang et al., 2017; Chung et al., 2021). In the final PV-recycling stage, the remaining materials after recycling are sent to landfills or incineration facilities.

In this study, a high-level EoL scenario was designed to analyze the environmental impact of the PV EoL treatment process, mainly composed of thermal treatment and chemical treatment methods based on the available life-cycle inventory from other literature. The conventional recycling process, which was adopted by European recyclers (Frischknecht et al., 2020), was proposed as a low-level EoL scenario, which was designed under two scenarios with different ultimate disposal methods: landfilling or incineration.

3.2.3.1. High-level EoL scenario (Scenario 1)

For the high-level scenario, the process first starts with transporting PV waste from discharge points to the recycling facility. Then, the c-Si PV panel requires manual disassembly of the aluminum frame and junction box. The junction box is separated into copper wire and polymers. The next step is removing the EVA layer to separate the glass from the silicon cell using thermal treatment. The outputs of this process are glass scrap and PV sandwich ashes, which are sent back to the metal recovery process. The polymers separated from the junction box, glass waste, fly ash, and sludge from the metal recovery process are sent to landfills. Figure 3.3 shows this process.

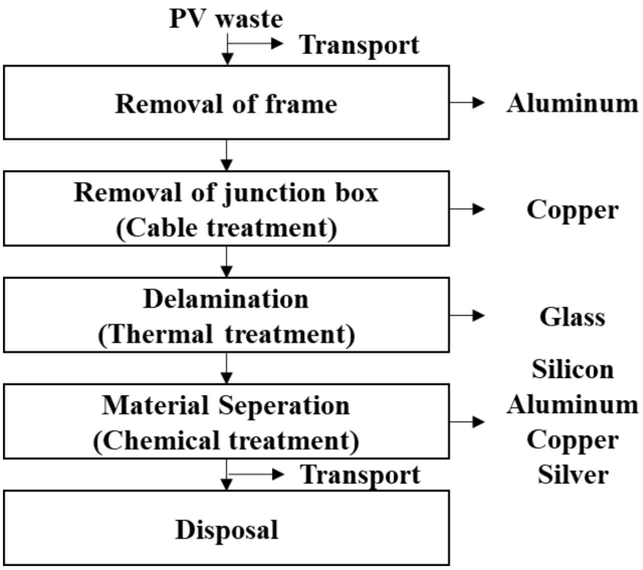


Fig. 3.3. Schematic diagram of high-level scenario: Scenario 1.

3.2.3.2. Low-level EoL scenario (Scenarios 2 and 3)

Analogous to Scenario 1, low-level scenarios began with transporting PV wastes from the discharge points. The junction box and aluminum frame were first separated. The residual PV cell scraps are shredded, followed by landfilling all residues in Scenario 2. Compared to Scenario 2, incineration replaces landfilling in Scenario 3. Figures 3.4 and 3.5 illustrate the schematic diagram of the low-level scenarios.

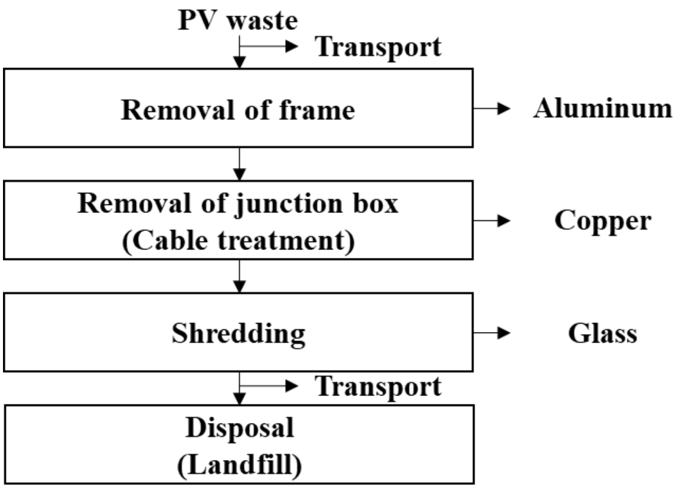


Fig. 3.4. Schematic diagram of low-level-landfill scenario: Scenario 2.

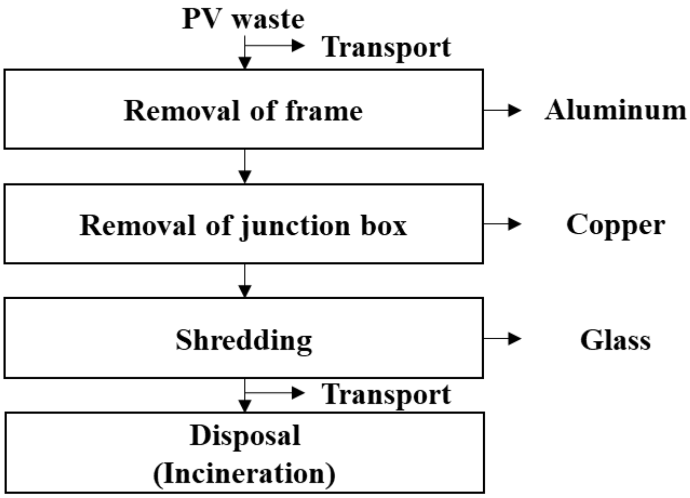


Fig. 3.5. Schematic diagram of low-level-incineration scenario: Scenario 3.

3.2.4. Life cycle inventory

Tables 3.2, 3.3, 3.4, and 3.5 show the LCIs of each process. All inventories are referred to as 1 ton of c-Si PV waste (functional unit). The life cycle inventory data for each process and material were derived from the Ecoinvent 3.5 database. The datasets in Ecoinvent were selected to obtain the best geographical and technological representativeness. With the exception of electricity consumption, the datasets with the "rest-of-the-world" or "global" inventory in Ecoinvent were used. The inventory used in this study did not include the datasets that reflected the national LCI database because of the limited information.

3.2.4.1. Life cycle inventory of high-level EoL treatment

In this study, it is assumed that the transport distance from the PV waste origin to a recycling facility is 200 km. This was approximated as an average distance between the regional collection center (the origin point) and recycling facility located at Jincheon-gun, Chungchenbuk-do, where a solar recycling center is under construction in Korea (Kim et al., 2019). It is also assumed that PV waste is transported using a 10-ton lorry from the origin point to a recycling facility.

When PV waste was unloaded using a wheel loader, the diesel consumption was assumed to be 64.8 MJ per 1 ton of PV waste (Stolz and Frischknecht, 2018). The electricity usage for removing the aluminum frame and junction box was assumed to be 5.3 kWh (Latunussa et al., 2016). Because the EVA and back-sheet layer (polyethylene terephthalate (PET) and PVF) were presumed to be incinerated during the thermal treatment step, inventory data for incinerating plastic mixtures, polyethylene, and polyvinyl fluoride were used, respectively. The input amount was based on the PV panel's composition (Table 3.1).

The inventory data of the material recovery process (acid leaching), such as the consumed nitric acid (HNO_3) and calcium hydroxide ($Ca(OH)_2$) amounts and the emission of NOx, were calculated using FRELP. The residuals from the recycling process were assumed to be transported to a sanitary landfill 150 km away using a 5-ton lorry for final disposal. The inventory data for landfilling glass waste were adopted for landfilling the inert waste, limestone sludge as limestone residue, sludge with metal residuals as sludge, and pig iron. Table 3.2 shows the LCI of each process.

3.2.4.2. Life cycle inventory of low-level EoL treatment

In the low-level EoL treatment process, the assumptions made for the inventory data set, electricity and diesel consumption for disassembly, and transport distances were the same as those in the high-level EoL treatment process. The mechanical treatment process was assumed to be accomplished by shredding using electricity, and its consumption was based on the previous report (Stolz and Frischknecht, 2018). In Scenario 2, the residuals of PV cell waste were presumed to be disposed of in a sanitary landfill. The inventory data for landfilling glass wastes was adopted for landfilling the glass waste to sanitary landfill, EVA, and back-sheet layer as a waste plastic mixture. Because the inventory data for silicon and metal waste subjected to sanitary landfill was unavailable in Ecoinvent 3.5, the emissions were calculated using the inventory data for consumer electronics waste.

In Scenario 3, the residuals of PV cell waste were assumed to be disposed of in the incineration facility. The inventory data for the incineration of glass waste was adopted for the incineration of waste glass, EVA as a waste plastic mixture, and PVF in the back-sheet layer as PVF waste. The inventory data for silicon and metal waste to incineration was adapted from that for the incineration of copper waste by modifying the emission data in proportion to the composition of PV waste.

3.2.4.3. Life cycle inventory of materials recovery

Based on the previous literature (Deng et al., 2018; Duflou et al. 2018; Maani et al., 2020; Latunussa et al., 2016), the material recovery yields for the high-level recycling were assumed to be 98% for glass, 100% for aluminum frame, 100% for copper from cables, 95% for silicon, 50% for aluminum, 95% for copper, and 94% for silver. It also was assumed that the recovered glass avoided the net consumption of the primary materials of silica sand, lime, and soda and the electricity during the foam glass production with and without glass cullet (Stolz and Frischknecht, 2018). The scraps of aluminum and copper from the frame and the cables were assumed to be treated further for the production of secondary materials. Since the supply mix of aluminum and copper holds the recycled contents, the avoided amount from the material recovery process was calculated by using the inventory of each primary material, i.e., 26% of the supply mix of aluminum cast alloy as a primary aluminum and 60% of the supply mix of copper as a primary copper (Stolz and Frischknecht, 2018). In the low-level EoL scenario, the material recovery yields were presumed to be 90% for glass and copper and 100% for aluminum, according to the previous literature (Stolz and Frischknecht, 2018; Duflou et al., 2018).

Table 3.2. Life cycle inventory of recycling of c-Si PV waste in high-level EoL treatment.

Process	Flow	Amount	Unit	Description/Assumption
Transportation	Transport, freight, lorry3.5–7.5 metric ton, EURO6	1000 × 200	kg × km	Assuming the distance (200 km) to the recycling site from PV waste origin using a 5-ton lorry
Disassembly	Diesel, burned in building machine	64.8	MJ	Diesel consumption for wheel loaders (data source: Stolz et al., 2018)
	Electricity, medium voltage/KR	5.3	kWh	Electricity consumption for disassembling (data source: Latunussa et al., 2016)
Delamination (thermal treatment)	Treatment of waste plastic, mixture to municipal incineration	74.1	kg	Combustion of EVA layer at the thermal treatment facility. The input amount was based on the PV panel's composition using LCI for municipal incineration of mixture plastic
	Treatment of waste polyethylene to municipal incineration	25.9	kg	Combustion of back-sheet layer at the thermal treatment facility. The input amount was based on the PV panel's composition using LCI for municipal incineration of mixture plastic
	Treatment of waste polyvinyl fluoride to municipal incineration	8.4	kg	Combustion of back-sheet layer at the thermal treatment facility. The input amount was based on the PV panel's composition using LCI for municipal incineration of mixture plastic
Material separation (chemical treatment)	Electricity, medium voltage/KR	87.3	kWh	Electricity consumption for chemical treatment (data source: Latunussa et al., 2016)
	Nitric acid, without water, in 50% solution state	10.9	kg	Acid leaching agent (data source: Latunussa et al., 2016)

	Lime, hydrated, loose weight	56.2	kg	Neutralization agent (data source: Latunussa et al., 2016)
	Water completely softened from decarbonized water at user	476.5	kg	Water consumption of acid leaching, electrolysis, neutralization (data source: Latunussa et al., 2016)
	Nitrogen oxides emission	3.1	kg	Emission of chemical treatment (data source: Latunussa et al., 2016)
Transportation	Transport, freight, lorry7.5–16 metric ton, EURO6	562.5 × 150	kg × km	Assuming the distance (150 km) to the landfill from the recycling facility using a 10-ton lorry. Disposal amount constitutes glass waste, ash waste of thermal treatment, and sludge/liquid waste of chemical treatment
Disposal	Average incineration residue	3.1	kg	Landfilling of ash
	Inert waste	13.2	kg	Landfilling of glass waste
	Limestone residue	469.2	kg	Landfilling of sludge waste
	Sludge, pig iron production	77	kg	Landfilling of liquid waste

Table 3.3. Life cycle inventory of recycling of c-Si PV waste in low-level EoL treatment ending up in a landfill.

Process	Flow	Amount	Unit	Description/Assumption
Transportation	Transport, freight, lorry3.5–7.5 metric ton, EURO6	1000 × 200	kg × km	Assuming the distance (200 km) to the recycling site from the PV waste origin using a 5-ton lorry
Disassembly	Diesel, burned in building machine	64.8	MJ	Diesel consumption for wheel loaders (data source: Stolz et al., 2018)
	Electricity, medium voltage/KR	5.3	kWh	Electricity consumption for disassembling (data source: Latunussa et al., 2016)
Material separation (Shredding)	Electricity, medium voltage/KR	111	kWh	Electricity consumption for disassembling (data source: Latunussa et al., 2016)
Transportation	Transport, freight, lorry7.5–16 metric ton, EURO6	242.1 × 150	kg × km	Assuming the distance (150 km) to the landfill from the recycling site using a 10-ton lorry. The disposal amount constitutes glass waste, ash waste of thermal treatment, and sludge/liquid waste of chemical treatment
Disposal	Treatment of waste glass, sanitary landfill	66.1	kg	Landfilling of waste glass
	Treatment of slag from silicon production	55.5	kg	Landfilling of silicon waste
	Treatment of waste plastic, mixture, sanitary landfill	120.5	kg	Landfilling of plastic waste (EVA, back-sheet layer, and others)

Table 3.4. Life cycle inventory of recycling of c-Si PV waste in the low-level EoL treatment ending up to be incinerated.

Process	Flow	Amount	Unit	Description/Assumption
Transportation	Transport, freight, lorry3.5–7.5 metric ton, EURO6	1000 × 200	kg × km	Assuming the distance (200 km) to the recycling site from the PV waste origin using a 5-ton lorry
Disassembly	Diesel, burned in building machine	64.8	MJ	Diesel consumption for wheel loaders (data source: Stolz et al., 2018)
	Electricity, medium voltage/KR	5.3	kWh	Electricity consumption for disassembling (data source: Latunussa et al., 2016)
Material separation (Shredding)	Electricity, medium voltage/KR	111	kWh	Electricity consumption for disassembling (data source: Latunussa et al., 2016)
Transportation	Transport, freight, lorry7.5–16 metric ton, EURO6	242.1 × 150	kg × km	Assuming the distance (150 km) to the landfill from the recycling site using a 10-ton lorry. The disposal amount constitutes glass waste, ash waste of thermal treatment, and sludge/liquid waste of chemical treatment
Disposal	Treatment of waste glass	66.1	kg	Incineration of waste glass
	Treatment of waste plastic, mixture	167.6	kg	Incineration of landfilled plastic waste (EVA, silicon, and others)
	Treatment of waste PVF	8.4	kg	Incineration of PVF in the back-sheet layer

Table 3.5. Life cycle inventory of the avoided burdens due to materials recovered from c-Si PV waste in high-level EoL scenario.

Process	Flow	Amount	Unit	Description/Assumption
Glass recovery	Electricity, medium voltage/KR	−244.1	kWh	Avoided primary glass production materials by recycled glass cullet Recycled glass cullet substitutes primary materials in foam glass production based on calculations with and without glass cullet
	Lime	−121.5	kg	
	Silica sand	−363.7	kg	
	Soda ash	−144.0	kg	
Aluminum recovery	Aluminum, cast alloy	−159.5	kg	Avoided aluminum scrap production from aluminum scrap by disassembling the frame
	Primary aluminum	−1.6	kg	Avoided primary aluminum materials recovered by chemical treatment
	Treatment of aluminum scrap	159.5	kg	Treatment of aluminum scrap from recycled aluminum frame
Copper recovery	Treatment of copper scrap	3.3	kg	Treatment of copper scrap from junction box and copper cables
	Market for copper	−10.6	kg	Avoided primary copper production materials from copper scrap and recovered copper by chemical treatment
Silicon recovery	Silicon, metallurgical grade	−52.7	kg	Avoided primary silicon production materials from recovered silicon by chemical treatment
Silver recovery	Market for silver	−0.2	kg	Avoided primary silver production materials from recovered silicon by chemical treatment

Table 3.6. Life cycle inventory of the avoided burdens due to materials recovered from c-Si PV waste in low-level EoL scenario.

Process	Flow	Amount	Unit	Description/Assumption
Glass recovery	Electricity, medium voltage/KR	−244.1	kWh	Avoided primary glass production materials by recycled glass cullet Recycled glass cullet substitutes primary materials in foam glass production based on calculations compared with foam glass production by glass cullet to foam glass production without cullet
	Lime	−121.5	kg	
	Silica sand	−363.7	kg	
	Soda ash, light, crystalline, heptahydrate	−144.0	kg	
Aluminum recovery	Aluminum, cast alloy	−159.5	kg	Avoided aluminum scrap production from aluminum scrap by disassembling the frame
	Treatment of aluminum scrap	159.5	kg	Treatment of aluminum scrap from recycled aluminum frame
Copper recovery	Treatment of copper scrap	3.3	kg	Treatment of copper scrap from junction box and copper cables
	Market for copper	−10.6	kg	Avoided primary copper production materials from copper scrap

3.2.5. Life Cycle Impact Assessment (LCIA)

The environmental assessment of two scenarios was accomplished by means of LCA open-source software, OpenLCA (ver. 1.10.2) (Winter et al., 2014) with the Ecoinvent v3.5 database. ReCiPe is a life cycle impact assessment (LCIA) methodology to reach a consensus on the recommended method for each environmental theme, at both the midpoint and the endpoint levels (Goedkoop et al., 2009). The eighteen midpoint indicators as shown in Table 3.7 for LCIA. Among them, the commonly used 11 midpoint categories were included in this study. The endpoint results are expressed in points (Pt) by transforming the value of environmental impacts into three endpoint indicators of damage (i.e., human health, ecosystem quality, and resource scarcity) at the weighting stage. These indicators represent a relative environmental impact and scores to support the interpretation of the LCA results. The midpoint analysis is accurate and has a strong relation to environmental effects, whereas the endpoint analysis is easier to interpret. In this study, the midpoint analysis was performed to evaluate the environmental impacts, in addition to the endpoint analysis for easier analysis.

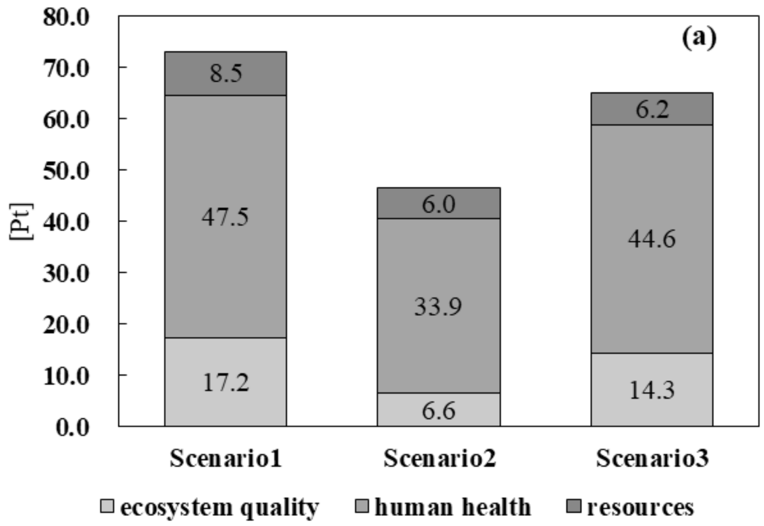
Table 3.7. Life cycle impact indicators (midpoint) used in this study.

Midpoint impact category	**Unit**	**Abbreviation**	**Used in this study**
Climate change	kg CO_2-Eq	GWP	O
Ozone depletion	kg CFC-11-Eq	ODP	O
Terrestrial acidification	kg SO_2-Eq	TAP	O
Freshwater eutrophication	kg P-Eq	FEP	O
Marine eutrophication	kg N-Eq	MEP	
Human toxicity	kg 1,4-DCB-Eq	HTP	O
Photochemical oxidant formation	kg NMVOC	POFP	O
Particulate matter formation	kg PM10-Eq	PMFP	
Terrestrial ecotoxicity	kg 1,4-DCB-Eq	TETP	O
Freshwater ecotoxicity	kg 1,4-DCB-Eq	FETP	O
Marine ecotoxicity	kg 1,4-DCB-Eq	METP	O
Ionising radiation	kg U235-Eq	IRP_HE	
Agricultural land occupation	$m^2 \times$ year	ALOP	
Urban land occupation	$m^2 \times$ year	ULOP	
Natural land transformation	m^2	NLTP	
Water depletion	m^3	WDP	
Metal depletion	kg Fe-Eq	MDP	O
Fossil depletion	kg oil-Eq	FDP	O

3.3. Results and discussion

3.3.1. Impact analysis results of EoL treatment of c-Si PV waste

The results of the endpoint analysis revealed that the highest and lowest environmental impact was caused by the EoL treatment processes of Scenarios 1 and 2, respectively (Fig. 3.6 (a)). These results were caused by higher environmental burdens associated with any processes requiring chemicals, electricity, and thermal treatment. In Scenario 1, the highest contributing process step was observed in the delamination and material recovery steps, whereas it was the final disposal in Scenario 3 (Fig. 3.6 (b)). The contribution percentages of disassembly, transport, delamination, and disposals in the three scenarios are in the range of 2%–4%, 16%–23%, 30%–41%, and 11%–60%, respectively.



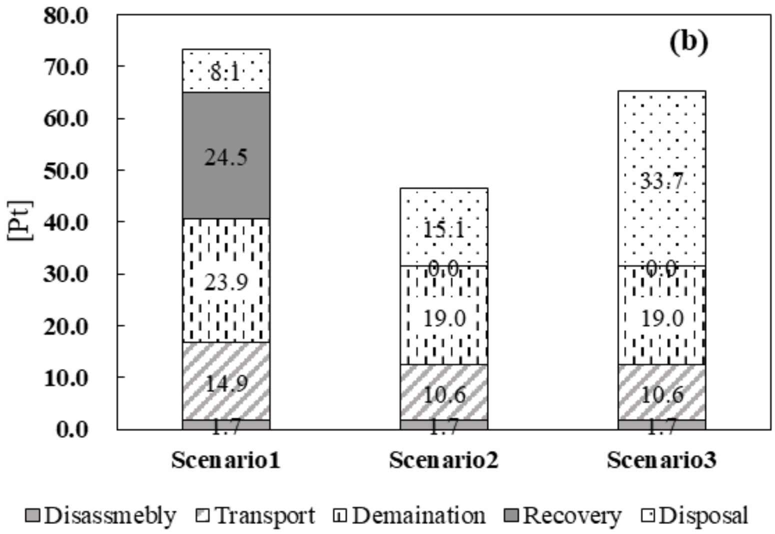


Fig. 3.6. Environmental impacts of 1 ton of PV EoL treatment process for the three scenarios using the ReCiPe endpoint method (a) by impact category and (b) process step.

Table 3.8 shows the environmental impact results for 1 ton of EoL PV treatment with the three scenarios using the ReCiPe midpoint method. Figure 3.7 shows the contribution of each step during the treatment, with relative values to the highest impacts scaled to 100%. By excluding three toxicity indicators (HTP, METP, and FETP), Scenario 1 shows the highest environmental impacts, whereas Scenario 2 was the lowest among the three scenarios. In the case of FETP and METP, the environmental impacts of Scenario 2 were the highest. Regarding the HTP indicator, the highest environmental impact was derived from Scenario 3.

Table 3.8. Environmental impacts for 1 ton of EoL PV treatment under the three scenarios using the ReCiPe midpoint method.

Category	Unit	Scenario 1	Scenario 2	Scenario 3
GWP	kg CO_2-Eq	5.23E+02	1.45E+02	3.98E+02
ODP	kg CFC-11-Eq	2.72E-05	1.80E-05	1.86E-05
TAP	kg SO_2-Eq	3.04E+00	4.73E-01	5.31E-01
FEP	kg P-Eq	8.90E-02	5.90E-02	6.06E-02
HTP	kg 1,4-DCB-Eq	4.01E+03	4.22E+03	4.53E+03
POFP	kg NMVOC	3.82E+00	4.50E-01	5.18E-01
TETP	kg 1,4-DCB-Eq	5.78E-01	3.28E-01	5.27E-01
FETP	kg 1,4-DCB-Eq	3.10E+00	9.18E+02	7.47E+02
METP	kg 1,4-DCB-Eq	3.31E+03	2.36E+05	1.93E+05
MDP	kg Fe-Eq	7.69E+00	3.78E+00	4.30E+00
FDP	kg oil-Eq	6.82E+01	4.85E+01	4.99E+01

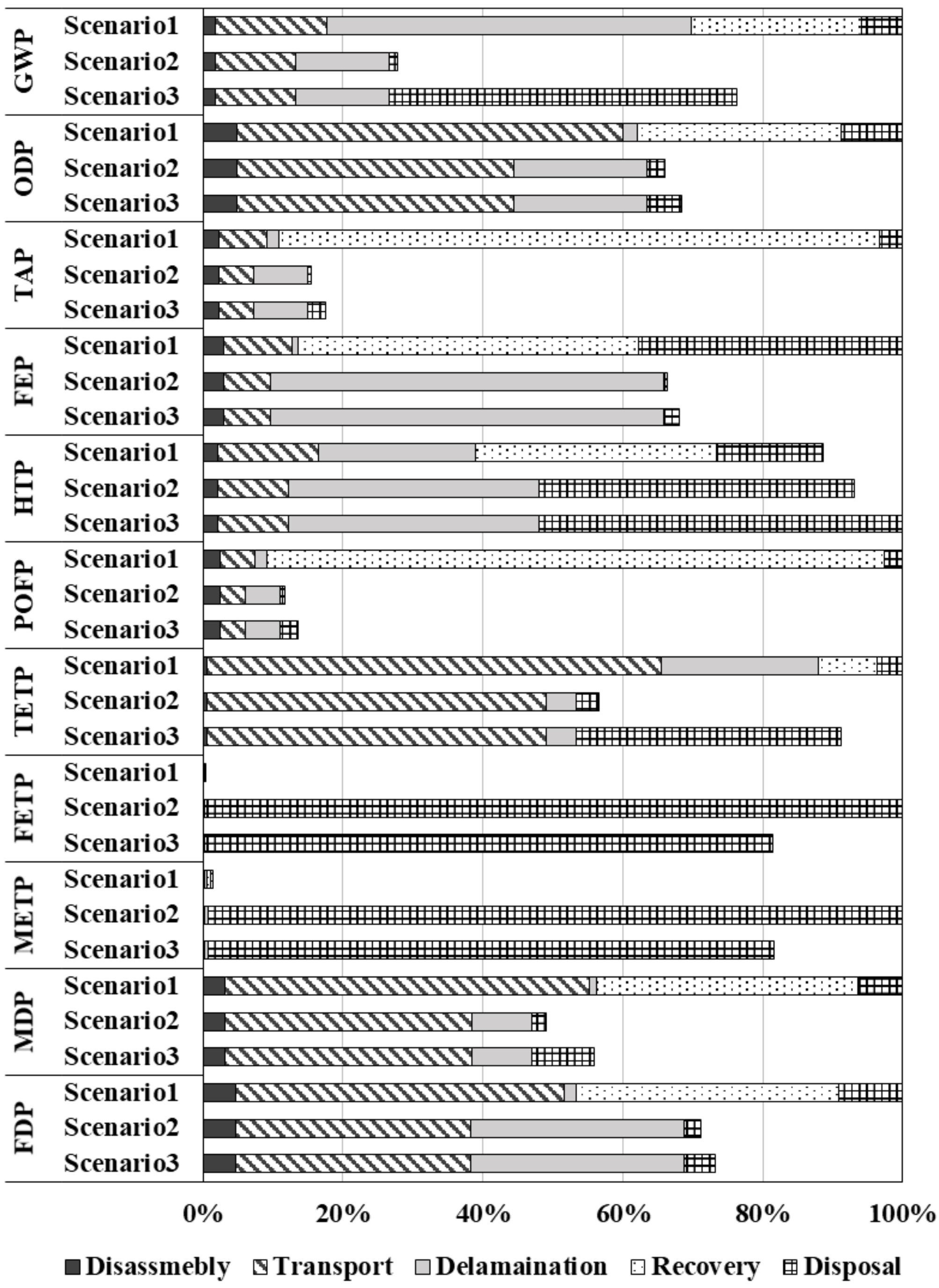


Fig. 3.7. Relative environmental impact results of the three scenarios for each midpoint indicator% by individual process step; the maximum result is set to 100.

With respect to GWP, the overall impact of Scenario 1 was around 523 kg CO_2 eq. This amount resulted mostly from delamination (52%), material recovery treatment (24%), and transport (16%). The delamination step mostly contributed to the overall impact of Scenario 1, which was caused by the emission of CO_2 during the incineration of the plastics, such as EVA, PET, and PVF of the back-sheet layer. The second highest GWP impact of Scenario 1 was caused mainly by the consumption of electricity and the neutralization agent used during the recovery treatment. The GWP impacts of low-level recycling scenarios were calculated as 145 and 398 kg CO_2 eq for Scenarios 2 and 3, respectively. In Scenario 3, 65% of the GWP impact was associated with the final disposal step with the incineration option and the emission of CO_2.

In the case of ODP and FDP, transportation contributed more than 50% of the overall impacts because of the use of diesel fuel for transportation. The electricity consumed for the material recovery in Scenario 1 was the second significant process that contributed to the ODP and FDP. Regarding the indicator of FEP, the electricity consumption during material recovery mainly contributed around 50% of the total impacts of Scenario 1. However, approximately 90% of the total impacts were related to the delamination step (shredding) in Scenarios 2 and 3.

In the cases of TAP and POFP, the material recovery process contributed to more than 90% to the overall impact of Scenario 1 because of NO_x emissions during the electrolysis process caused by using nitric acid as a leaching agent in the material recovery step.

The environmental impact by electricity consumption for the material recovery process in Scenario 1 contributed more than 30% HTP. In Scenario 1, 25% of HTP impact was generated by incinerating plastic components (EVA, PET, and PVF) during the thermal treatment step. The HTP impact of Scenarios 2 and 3 is higher than that of Scenario 1 because of hazardous metal emissions (lead, silver ion, and copper ion) to the groundwater during the final disposal step.

For the TETP impact category, transportation contributed more than 50% in each scenario. Brake abrasion from road freight transportation contributed about 80% of the TETP impact. The transport step contributed more than 50% of the MDP impact to the overall impacts associated with producing vehicles. The second significant process that contributed to the MDP impact category was HNO_3 production in Scenario 1.

Regarding the FETP and METP impact categories, the impacts of Scenarios 2 and 3 were almost 300 times higher than the impact of Scenario 1. The emission of copper ions and silver ions to the groundwater during the final disposal step was found to be the most significant contributor to these impacts. Because the emission inventory for the residuals after the material recovery process in Scenario 1 is unavailable, the inventory of the pig iron sludge was used in this study.

3.3.2. Impact analysis results of environmental benefits of material recovery

In this study, the environmental benefits associated with the recovery of materials were considered separately from the environmental impact of EoL treatment. The secondary materials are produced by the recycling of the recovered materials. These materials are assumed to be suitable as substitutes for primary materials, thereby avoiding the production of the primary materials. The potential benefits are accounted as the net impacts of the credits due to the avoided environmental impacts caused by using primary materials that can be replaced by secondary materials and the burdens caused by the production of secondary materials (Stolz and Frischknecht, 2018; Ardente et al., 2019).

Analyzing the environmental benefits for the recovered materials was performed considering the amount of secondary materials produced, considering recycling yields for each scenario. The recovered materials are aluminum, glass, copper, silicon, and silver for Scenario 1 and aluminum, glass, and copper for Scenarios 2 and 3. The recovered material amount is the same for Scenarios 2 and 3.

Figure 3.8 (a) shows the ReCiPe endpoint impacts of material recovery with the impact of EoL treatment. The negative numbers of impacts indicate that the environmental impacts of producing primary materials are higher than those of producing secondary materials because of the EoL treatment process. The results showed that the environmental benefits of Scenario 1 were approximately two times higher than those of Scenarios 2 and 3 because of the highly recovered materials by additional EoL treatment processes.

In all scenarios, the highest contribution to the environmental benefit was observed from copper recovery (Fig. 3.6 (b)). The contribution percentage was calculated to be 46% and 60% of the overall environmental benefits for Scenario 1 and Scenarios 2 and 3, respectively. In Scenario 1, the contribution percentage of silver and aluminum was 19% and 15%, respectively. These results were different from those of previous literature that reported that the contribution of aluminum recovery was the most significant factor for environmental benefits (Ardente et al., 2019; Dias et al., 2021). This can be explained because the literature has assumed that all amounts of the recovered aluminum substitute the primary aluminum, whereas this study has considered only the amount of primary material that can be shared in the supply mix to be displaced by recycled material (Stolz and Frischknecht, 2018). The environmental benefit linked to the recycled aluminum was lower than copper because the content of the primary material in the supply mix of aluminum and copper was 26% and 60%, respectively.

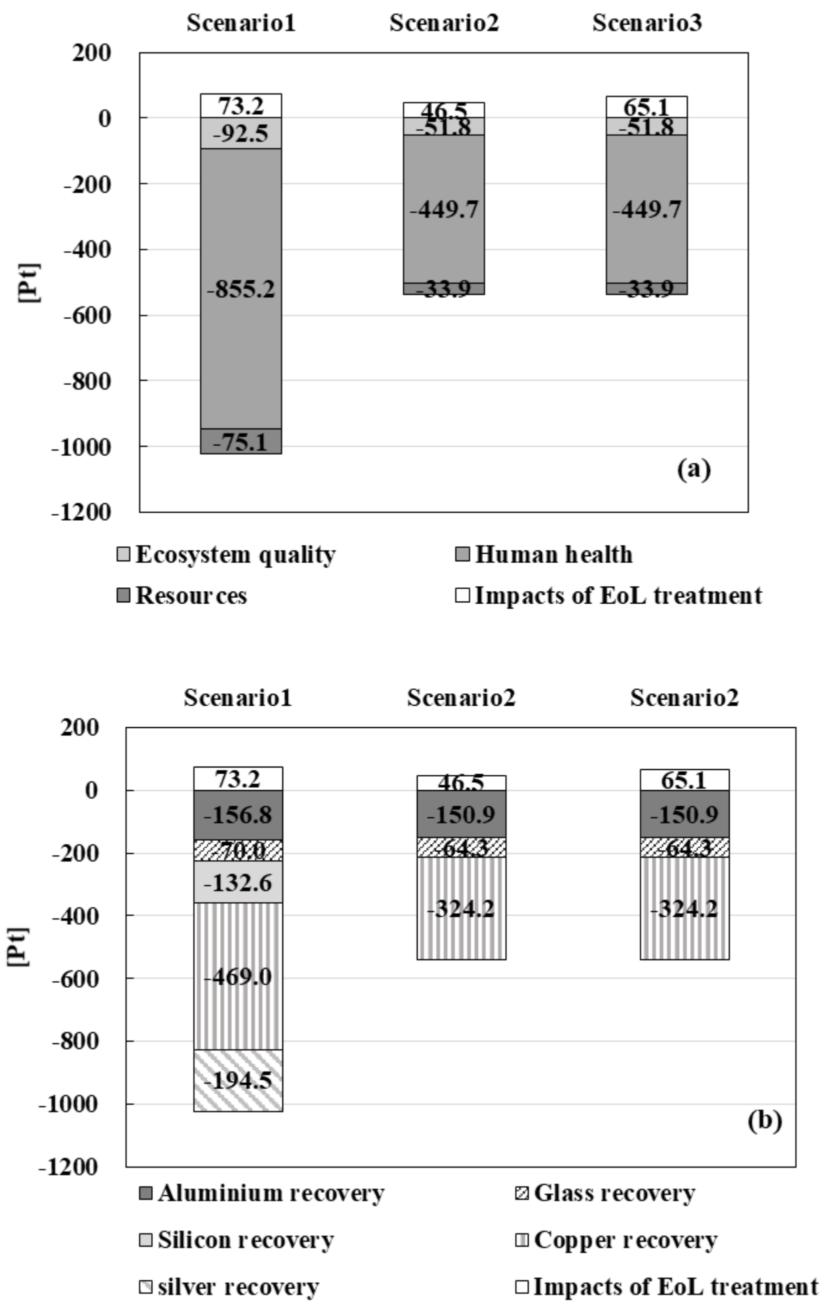


Fig. 3.8. Environmental burdens and benefits of 1 ton of PV recycling for three scenarios according to the ReCiPe endpoint method (a) by impact category and (b) recycling step.

Figure 3.9 illustrates the contribution of each recovered material to the overall environmental burdens and benefits analyzed using ReCiPe midpoint indicators. Regarding the impact categories of GWP, FETP, METP, and FDP, the aluminum recovery was the main contributor to the overall environmental benefits because of the reduced emissions during its primary production. The copper recovery could avoid the emission of toxic metals to groundwater during its primary production, exhibiting a prominent environmental benefit associated with HTP. During its production, the dominant contributor to TETP was related to copper emissions to the air. Even though only 0.5 kg of silver was recovered from 1 ton of PV waste, the environmental benefits by avoiding a primary raw material were positive for the impact categories of FET, HTP, METP, and MDP.

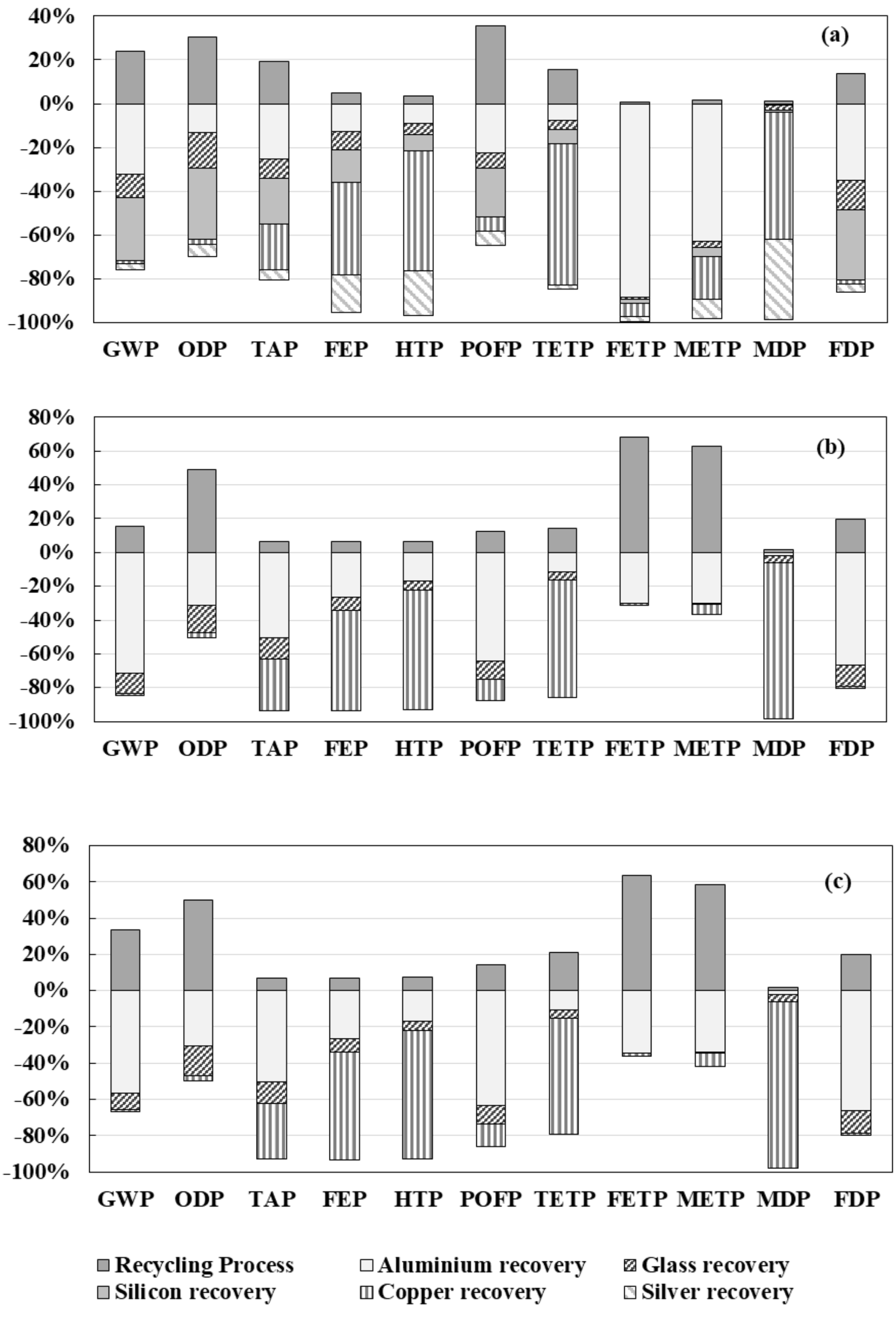


Fig. 3.9. Contribution of the material recovery to the overall environmental burdens and benefits of the recycling PV wastes according to ReCiPe midpoint method in (a) Scenario 1, (b) Scenario 2, (c) Scenario 3.

3.3.3. Total impacts of EoL scenarios of c-Si PV panel

In Table 3.9, the overall environmental impacts associated with EoL treatment, including the impacts from the material recovery, are shown according to the endpoint method. The scores (net benefits) of Scenario 1 were higher than they were for the other scenarios. The environmental benefit due to the recovery of material was fourteen times higher than the environmental impact by EoL treatment in Scenario 1, whereas it was 8.9 and 7.8 times higher in Scenarios 2 and 3, respectively.

Table 3.10 shows the results for the ReCiPe midpoint method. Except for the FETP and METP of Scenario 2 and 3, together with the ODP of Scenario 3, the environmental benefits exceeded the environmental burdens by factors of 2 to 146 for Scenario 1 and by 1 to 15 for Scenarios 2 and 3.

The analysis was performed to assess the significance of EoL treatment of PV waste when compared to the production of the PV panel. The impact of the production was analyzed based on the inventory data from the literature (Frischknecht et al., 2020) and the Ecoinvent database.

Figure 3.10 and Table 3.11 show that the environmental burdens caused by the EoL treatment in Scenario 1 account for only 1.4%, compared to those by PV panel production. In Scenario 1, compared to PV panel production, the net environmental benefit generated by recycling recovered materials was estimated to be 19.2%, whereas that of Scenarios 2 and 3 was only 9.7% and 9.6%, respectively. When substituting the primary material input with secondary production for the PV panels' aluminum frame, the environmental burden can be avoided by ~50%, whereas the other half is associated with treating the recovered Al scrap. However, 84% of the environmental burden can be avoided by recycling copper as a secondary cable material.

Although the assessment of impact of the entire life cycle stage of the panel, including production, use, and installation, is not part of the scope of this study, the comparison of the impacts related to the EoL stage with the production stage can be useful for estimating the environmental perspectives through the entire life of the PV panel.

Table 3.9. Total environmental impacts for 1 ton of EoL PV treatment under the three scenarios using ReCiPe endpoint method.

Impact category	Unit	Scenario1			Scenario2			Scenario 3		
		Impact of EoL treatment	Impact of material recovery	Total impact	Impact of EoL treatment	Impact of material recovery	Total impact	Impact of EoL treatment	Impact of material recovery	Total impact
Ecosystem quality	**Pt**	17.2	−92.5	−75.3	6.6	−51.8	−45.2	14.3	−51.8	−37.5
Human health	**Pt**	47.5	−855.2	−807.7	33.9	−449.7	−415.8	44.6	−449.7	−405.1
Resources	**Pt**	8.5	−75.1	−66.6	6.0	−33.9	−27.9	6.2	−33.9	−27.7
Total Impacts	**Pt**	73.2	−1022.8	−949.7	46.5	−535.4	−488.9	65.1	−535.4	−470.3

Table 3.10. Total environmental impacts for 1 ton of EoL PV treatment under the three scenarios using ReCiPe midpoint method.

Impact category	Unit	Scenario1			Scenario2			Scenario 3		
		Impact of EoL treatment	Impact of material recovery	Total impact	Impact of EoL treatment	Impact of material recovery	Total impact	Impact of EoL treatment	Impact of material recovery	Total impact
GWP	kg CO2-Eq	5.23E+02	−1.65E+03	−1.13E+03	1.45E+02	−7.98E+02	−6.53E+02	3.98E+02	−7.98E+02	−4.00E+02
ODP	kg CFC-11-Eq	2.72E-05	−6.30E-05	−3.58E-05	1.80E-05	−1.85E-05	−5.50E-07	1.86E-05	−1.85E-05	1.16E-07
TAP	kg SO2-Eq	3.04E+00	−1.26E+01	−9.57E+00	4.73E-01	−7.07E+00	−6.60E+00	5.31E-01	−7.07E+00	−6.54E+00
FEP	kg P-Eq	8.90E-02	−1.82E+00	−1.73E+00	5.90E-02	−8.35E-01	−7.76E-01	6.06E-02	−8.35E-01	−7.74E-01
HTP	kg 1,4-DCB-Eq	4.01E+03	−1.18E+05	−1.14E+05	4.22E+03	−5.85E+04	−5.43E+04	4.53E+03	−5.85E+04	−5.40E+04
POFP	kg NMVOC	3.82E+00	−6.93E+00	−3.11E+00	4.50E-01	−3.17E+00	−2.72E+00	5.18E-01	−3.17E+00	−2.65E+00
TETP	kg 1,4-DCB-Eq	5.78E-01	−3.13E+00	−2.56E+00	3.28E-01	−1.99E+00	−1.66E+00	5.27E-01	−1.99E+00	−1.46E+00
FETP	kg 1,4-DCB-Eq	3.10E+00	−4.53E+02	−4.50E+02	9.18E+02	−4.24E+02	4.94E+02	7.47E+02	−4.24E+02	3.23E+02
METP	kg 1,4-DCB-Eq	3.31E+03	−1.77E+05	−1.73E+05	2.36E+05	−1.38E+05	9.84E+04	1.93E+05	−1.38E+05	5.51E+04
MDP	kg Fe-Eq	7.69E+00	−5.51E+02	−5.43E+02	3.78E+00	−2.26E+02	−2.23E+02	4.30E+00	−2.26E+02	−2.22E+02
FDP	kg oil-Eq	6.82E+01	−4.22E+02	−3.53E+02	4.85E+01	−1.98E+02	−1.49E+02	4.99E+01	−1.98E+02	−1.48E+02

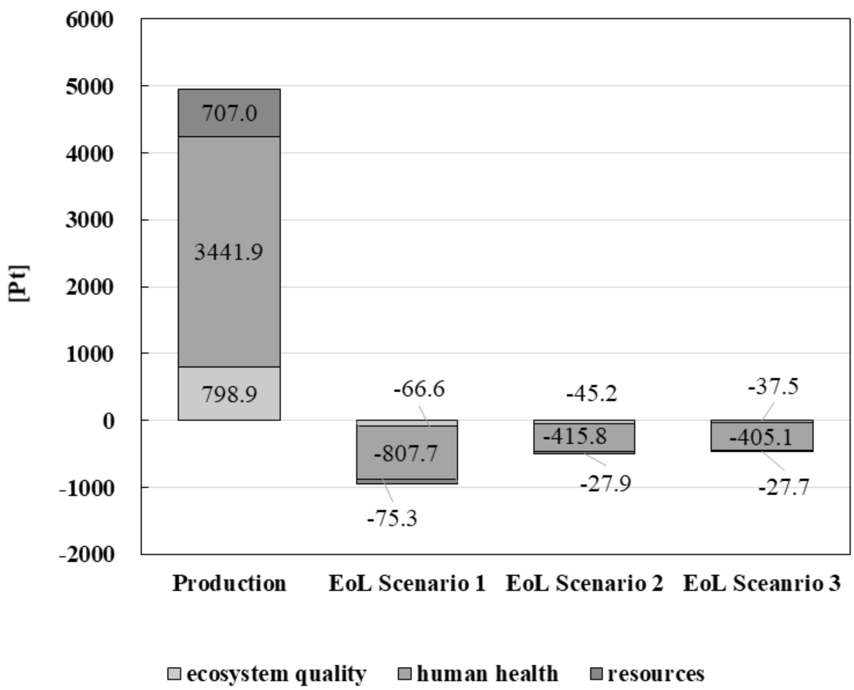


Fig. 3.10. Comparison of the environmental impacts between the production and each EoL scenario for 1 ton of PV panel.

Table 3.11. Total environmental impacts of the production and EoL scenarios of 1ton of PV panel.

Material/Process	Production	Scenario 1	Scenario 2	Scenario 3
Aluminum frame	284.7	−150.9	−150.9	−150.9
Glass	126.4	−70.0	−64.3	−64.3
Copper for cable	384.3	−324.2	−324.2	−324.2
PV cell (silicon for EoL)	3500.1	−132.6		
Other material/process	652.2	73.2	46.5	65.1

3.3.4. Comparative analysis of the variations with different parameters

Because the input parameters in this study are based on the literature and assumptions, factors linked to the actual EoL PV treatment can affect some parameters, which can influence environmental impacts. Here, comparative analysis has been performed to evaluate the relevance of the parameters to the impact results. For thermal treatment, the variability of data input parameters is ascribed to the difference in the amount of each plastic component. Table 3.12 shows the variability of each main component. The representative data used in this study have been adopted from the literature (Latunussa et al., 2016; Wambach et al., 2018; Maani et al., 2020; Duflou et al., 2018). Among ten categories, GWP, HTP, and TETP were the most prominent impact categories associated with thermal treatment. From the comparative analysis results, the relative impact was varied up to −20.0% for TETP, depending on the input amount of plastic components.

Table 3.12. Input parameters for the comparative analysis with respect to the plastic composition of PV waste.

Plastic component	Amount in this study (kg/1 ton)	Variability (kg/1 ton)	Data sources
EVA	74.1	57.5 - 74.1	Case A from Latunussa et al., (2016)
PVF	25.9	19.1 - 37.7	Case B from Wambach et al., (2018) Case C from Maani et al., (2020)
PET	8.4	8.0 - 15.0	Case D from Duflou et al., (2018)

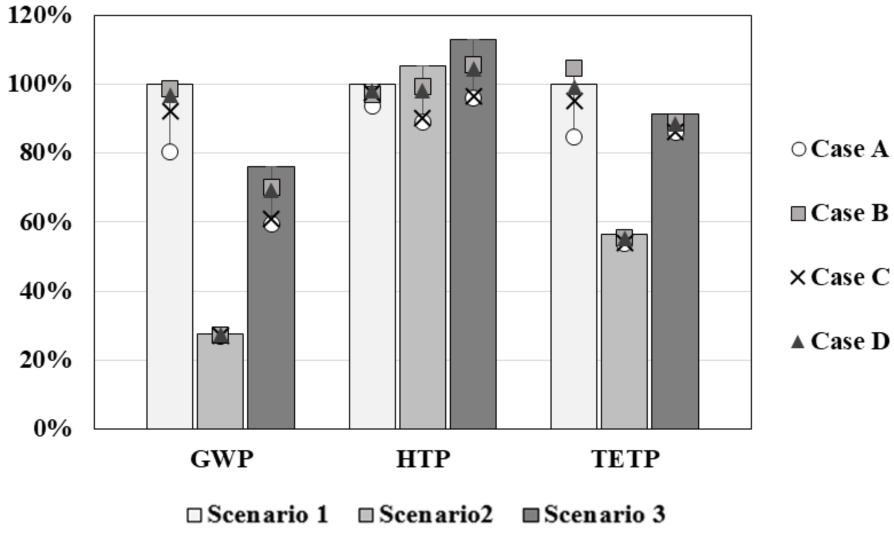


Fig. 3.11. Relative environmental impacts of the EoL treatment including the effect of the variance according to the plastic composition; the result of Scenario 1 is set to 100%.

Depending on the optimization degree, the process scale (industrial, pilot, or lab scale), and the detailed condition of chemical reactions, the number of input and output materials for the material recovery process has been varied (Huang et al., 2017; Latunussa et al., 2016; Dias et al., 2021). Particularly, the input amount of HNO3 differed by 2800 times from 0.03 kg/ton waste (Huang et al., 2017) to 84.68 kg/ton waste (Dias et al., 2021).

Figure 3.12 shows the influence of the comparative analysis for the material recovery process of Scenario 1. The nine categories contributing to the entire environmental impact by more than 10% were selected for comparative analysis. The influence of variable parameters related to the material recovery process for EoL treatment was in the range of −80% to +200%.

Because NOx emissions significantly influenced the impact categories of TAP and POFP, those impacts were lowered by more than 80% when the NO_x emissions equaled 0.16 kg/ton (Huang et al., 2017). Although the environmental impacts were drastically reduced in Scenario 1, the values were still higher than those of Scenarios 2 and 3 for all impact categories. The MDP influence varied from −50 to +200%, because of the difference in the input amount of HNO3.

Table 3.13. Parameters used for the comparative analysis with respect to the material recovery process.

Input/ *Output*	**Amount in this study (kg/1 ton)**	**Variability (kg/1 ton)**	Data source
HNO_3	10.9	0.30–84.68	Case A from Huang et al. (2017) Case B from Latunussa, et al. (2016) Case C from Dias, et al. (2021)
$Ca(OH)_2$	56.2	36.5–69	
Electricity	87.3*	56.76–100.81*	
Water	476.5	309.71–476.5	
NO_x	3.1	0.16–3.1	

* Unit : kwh/ton

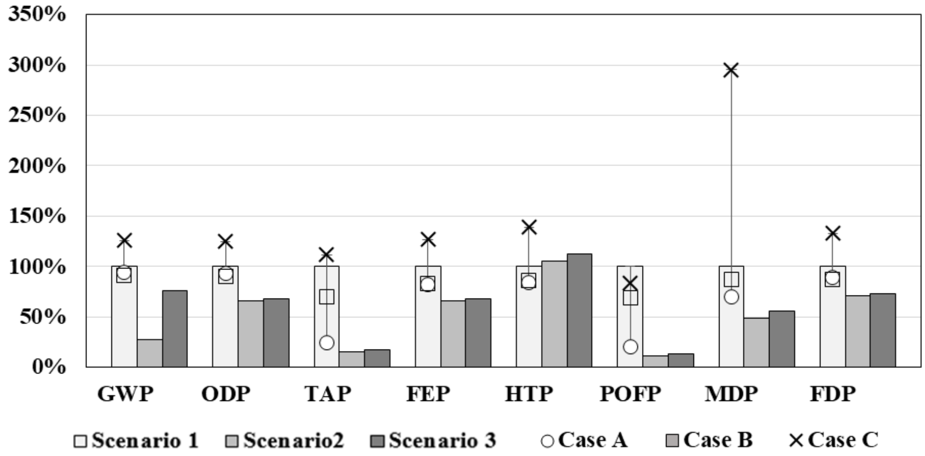


Fig. 3.12. Relative environmental impacts of EoL treatment including the effect of the variance according to the material recovery process; the result of Scenario 1 is set to 100%.

As mentioned earlier, the transport step contributed more than 50% to the impact categories of ODP, FDP, and TETP. The input parameters for the inventory of transport used in this study were the amount that was transported, the distance traveled, and the type of vehicle. The estimated distance and the type of lorry from previous studies are shown in Table 3.13. The results of the comparative analysis indicated that the lowest variance (-13 to +8%) was exhibited by the impact category of HTP, whereas the TETP caused the highest variance (-40 to +58%) among the environmental impact categories.

Table 3.14. Input parameters with variance according to transportation.

Input	Amount in this study (kg × km, lorry type)	Variability (kg × km, lorry type)	Data source
Transport 1	1,000 × 200, 3.5-7.5t-lorry	1,000 × (100 – 500), 3.5-7.5t-lorry, 16-32t-lorry	Case A : Latunussa, et al. (2016)
Transport 2	562.5 × 150, 7.5-16t-lorry	(disposed amount) × (0-100), 3.5-7.5t-lorry	Case B : Lunardi et al. (2018)

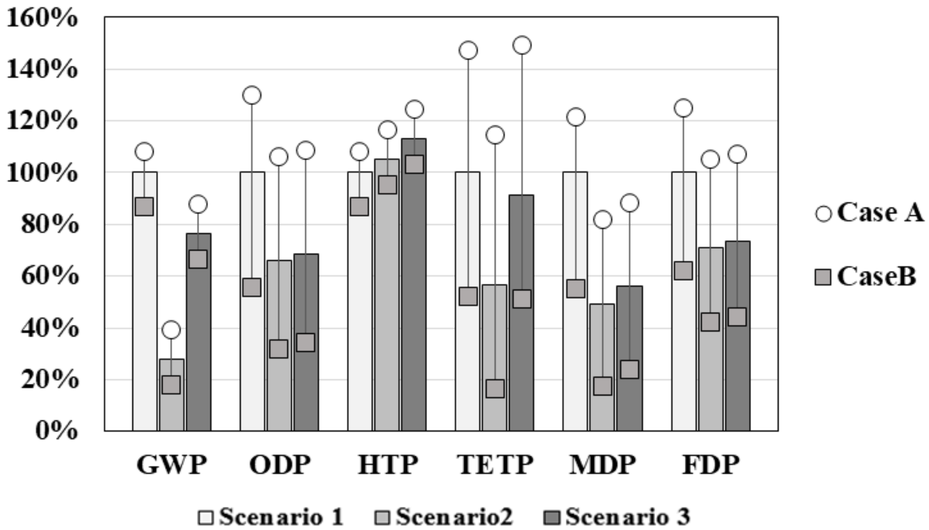


Fig. 3.13. Relative environmental impacts of EoL treatment including the effect of the variance according to the transportation; the result of Scenario 1 is set to 100%.

The input parameters regarding material recovery vary with the PV panel's composition, affecting the recoverable amount, whereas the recovered material amount depends on the recycling yield. Table 3.15 shows the variance of the recovered amount with different recycling yields obtained from previous studies for the same PV panel composition. The recycling yields were selected from the literature, including recycling methods, such as thermal delamination and chemical recovery, which could be classified as high-level recycling (Ardente et al., 2019; Maani et al., 2021; Duflou et al., 2018; Huang et al., 2017).

As shown in Fig. 3.14, three of four studies reported lower recycling yields than this study. For the environmental benefits provided by material recovery, the recycling yield's influence was in the range from −88% to +6% for all scenarios. For example, the environmental benefits of TETP, MDP, and HTP ranged from −88% to +6%, −71% to +4%, and −68% to +4%, respectively. These results depend on the recovered copper amount. The environmental benefits obtained in this study are noticeably different from those reported by Huang et al. (2017) because they excluded copper recycling from the EoL PV panels.

Table 3.15. Parameters used for the comparative analysis for the amount of recovered materials.

Recycling	**Recovered material**	**Quantity (kg/ton)**	**Variability (kg/ton)**	**Data source**
High-level recycling	Glass	648	595–648	Case A from Ardente et al. (2019); Case B from Maani et al. (2021); Case C from Duflou et al. (2018); Case D from Huang et al. (2017)
	Aluminum	161.1	132–162.7	
	Copper	10.6	0–10.9	
	Silicon	52.7	0–55.5	
	Silver	0.2	0–0.2	
Low-level recycling	Glass	595.1	29.8–595.1	Case E from Ardente et al. (2019); Case F from Stolz et al. (2016);
	Aluminum	159.5	151.5–159.5	
	Copper	3.3	3.2–3.3	

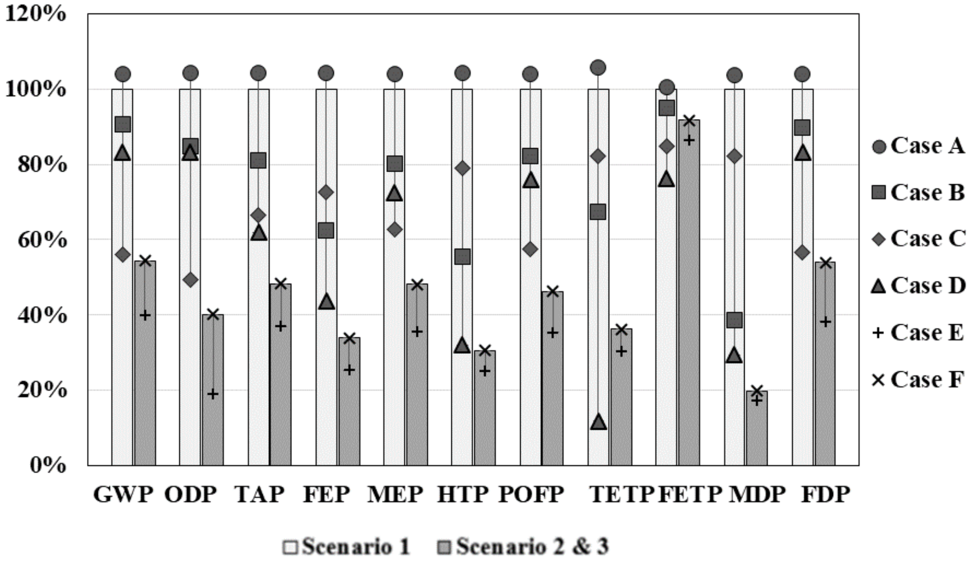


Fig. 3.14. Relative environmental benefits of the EoL treatment including the effect of variance according to the recycling yield; the result of Scenario 1 is set to 100%.

3.4. Summary

In this chapter, the environmental impacts associated with EoL c-Si PV panel treatment were compared and analyzed for three recycling methods while considering the practical applicability of EoL treatment in Korea. The LCA methodology was used for the analysis using the Ecoinvent 3.5 database and data available in the literature using the OpenLCA software.

The ReCiPe results revealed that the highest environmental impact was produced by an EoL treatment process of Scenario 1, but the lowest was caused by Scenario 2 for the overall impact categories at endpoint levels and midpoint levels except for a few indicators.

- In Scenario 1, the process with the highest contribution was the material recovery step because of the input of HNO_3 and the emissions of NO_x.
- Remarkably, the impacts of Scenarios 2 and 3 as much as almost 300 times higher than those of Scenario 1, mainly led by the FETP and METP impact categories because of the emission of copper and silver ions to the groundwater.
- The impact category of GWP was governed significantly by the emission of CO_2 during the incineration of the plastic component and the electricity consumption during the treatment process.
- With respect to the impacts of ODP, FDP, and TETP, transportation contributed more than 50% of the overall environmental burden.

The environmental benefits of Scenario 1 were approximately two times higher than those of Scenarios 2 and 3. The highest contribution to the environmental benefits was obtained from copper recovery using endpoint analysis.

- The aluminum recovery was the main contributor to the environmental benefits of the GWP, FETP, METP, and FDP impact categories.
- The copper and silver recovery exhibited a prominent environmental benefit associated with the toxicity impact categories (HTP, FETP, TETP, and METP).
- The content of the primary material in the material supply mix was the environmental benefits of recycling.
-

The environmental benefits of Scenario 1 were approximately two times higher than those of Scenario 2 and Scenario 3. The highest contribution to the environmental benefits was obtained from the recovery of copper according to the endpoint analysis.

- The Al recovery was found to be the main contributor to the environmental benefits of the impact categories of GWP, FETP, METP, and FDP.
- The copper and silver recovery exhibited a prominent environmental benefit associated with the toxicity impact categories (such as HTP, FETP, TETP, and METP).
- The content of the primary material in the material supply mix was found to be the environmental benefits of recycling.

The net benefits of Scenario 1 were higher than that of the other two scenarios. For FETP and METP of Scenarios 2 and 3 and the ODP of Scenario 3, the environmental burdens linked to the EoL treatment process were higher than the environmental benefits from the recovered materials. In Scenario 1, compared to PV panel production, the net environmental benefit generated by recycling recovered materials was estimated to be 19.2%, whereas that of Scenarios 2 and 3 was only 9.7% and 9.6%, respectively.

The variance of the parameters related to the material recovery process influenced the environmental impacts in the range of −80% to +200%. The recycling yields also governed the environmental benefits in the range of -88% to +6%. Depending on the variance of the parameter, such as traveling distance and route together with vehicle type, the environmental impacts associated with transportation were also influenced in the range of −40% to +58%, whereas the variance related to the delamination process ranged up to -20%.

References

Acero, A.P., Rodriguez, C., Ciroth, A., 2017. LCIA methods: Impact assessment methods in life cycle assessment and their impact categories. Version 1.5.6. Green Delta 1–23.

Ardente, F., Latunussa, C.E.L., Blengini, G.A., 2019. Resource efficient recovery of critical and precious metals from waste silicon PV panel recycling. Waste Manag. 91, 156–167.

Chung, J., Seo, B., Lee, J., Kim, J.Y., 2021. Comparative analysis of I2-KI and HNO3 leaching in a life cycle perspective: Towards sustainable recycling of end-of-life c-Si PV panel. J. Hazard. Mater. 404, 123989.

Corcelli, F., Ripa, M., Leccisi, E., Cigolotti, V., Fiandra, V., Graditi, G., Sannino, L., Tammaro, M., Ulgiati, S., 2018. Sustainable urban electricity supply chain – Indicators of material recovery and energy savings from crystalline silicon photovoltaic panels end-of-life. Ecol. Indic. 94, 37–51.

Deng, R., Chang, N.L., Ouyang, Z., Chong, C.M., 2019. A techno-economic review of silicon photovoltaic module recycling. Renew. Sustain. Energy Rev. 109, 532–550.

Dias, P., Schmidt, L., Monteiro, M., Chang, N.L., Spier, G., Corkish, R., Veit, H., 2021. Comprehensive recycling of silicon photovoltaic modules incorporating organic solvent delamination – technical , environmental and economic analyses. Resour. Conserv. Recycl. 165, 105241.

Duflou, J.R., Peeters, J.R., Altamirano, D., Bracquene, E., Dewulf, W., 2018. Demanufacturing photovoltaic panels: Comparison of end-of-life treatment strategies for improved resource recovery. CIRP Ann. 67, 29–32.

European Commission (EC), 2012. Directive 2012/19/EU of the European Parliament and of the Council of 4 July 2012 on waste electrical and electronic equipment (WEEE).

Fraunhofer ISE, 2020. Photovoltaics report, freiburg. Available at: ttps://www.ise. fraunhofer.de/content/dam/ise/de/documents/publications/studies/Photovoltaics-Report.pdf. (Accessed 7 June 2019).

Frischknecht, R.F; Itten, R.; Sinha, P.; de Wild-Scholten, M.; Zhang, M.; Fthenakis, V.; Kim, H.; Raugei, M.; Stucki, M., 2020. Life Cycle Inventories and Life Cycle Assessments of Photovoltaic Systems 2020 Task 12 PV Sustainability.

Goedkoop, M., Heijungs, R., Huijbregts, M., Schryver, A. De, Struijs, J., Zelm, R. Van, 2009. ReCiPe 2008 a life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. Report I: Characterisation.

Held, M., 2013. LCA screening of a recycling process for silicon based PV modules. PV Cycle Conf. 1–19.

Huang, B., Zhao, J., Chai, J., Xue, B., Zhao, F., Wang, X., 2017. Environmental influence assessment of China’s multi-crystalline silicon (multi-Si) photovoltaic modules considering recycling process. Sol. Energy 143, 132–141.

International Energy Agency (IEA), 2020. Methodology guidelines on Life Cycle Assessment of photovoltaic

International Organization for Standardization (ISO), Standard ISO 14040 Environmental management – Life cycle assessment – Principles and frame- work, 2006.

Kim, B., Kang, J., Park, S., Jang, J., Hong, J.H., 2019. Study of unit cost estimation for the appropriate treatment of end-of-life photovoltaic (PV) module. New & Renewable Energy 2019. 3 Vol. 15, No. 1

Klugmann-Radziemska, E., Ostrowski, P., Drabczyk, K., Panek, P., Szkodo, M., 2010. Experimental validation of crystalline silicon solar cells recycling by thermal and chemical methods. Sol. Energy Mater. Sol. Cells 94, 2275–2282.

Latunussa, C.E.L., Ardente, F., Blengini, G.A., Mancini, L., 2016. Life Cycle Assessment of an innovative recycling process for crystalline silicon photovoltaic c. Sol. Energy Mater. Sol. Cells 156, 101–111.

Lunardi, M., Alvarez-Gaitan, J., Bilbao, J., Corkish, R., 2018. Comparative Life Cycle Assessment of End-of-Life Silicon Solar Photovoltaic Modules. Appl. Sci. 8, 1396.

Maani, T., Celik, I., Heben, M.J., Ellingson, R.J., Apul, D., 2020. Environmental impacts of recycling crystalline silicon (c-SI) and cadmium telluride (CDTE) solar panels. Sci. Total Environ. 735, 138827.

Ministry of Environment (ME), 2019. Act on resource circulation of electrical and electronic equipment and vehicles. Republic of Korea.

Müller, A., Wambach, K., Alsema, E., 2005. Life Cycle Analysis of Solar Module Recycling Process. MRS Proc. 895, 0895-G03-07.

Paiano, A., 2015. Photovoltaic waste assessment in Italy. Renew. Sustain. Energy Rev. 41, 99–112.

Palitzsch, W., Schonherr, P., Loser, U., 2014. Integrated approach for economic PV waste recycling. 2014 IEEE 40th Photovolt. Spec. Conf. PVSC 2014 2709–2711.

Peeters, J.R., Altamirano, D., Dewulf, W., Duflou, J.R., 2017. Forecasting the composition of emerging waste streams with sensitivity analysis: A case study for photovoltaic (PV) panels in Flanders. Resour. Conserv. Recycl. 120, 14–26.

Santos, J.D., Alonso-García, M.C., 2018. Projection of the photovoltaic waste in Spain until 2050. J. Clean. Prod. 196.

Stolz, P., Frischknecht, R., 2018. Life Cycle Assessment of Current Photovoltaic Module Recycling.

Stolz, P., Frischknecht, R., Wyss, F., De Wild-Scholten, M., 2016. PEF screening report of electricity from photovoltaic panels in the context of the EU Product Environmental Footprint Category Rules (PEFCR) Pilots. Pvthin.Org.

Wambach, K., Sinha, P., Health, G., 2018. Life Cycle Inventory of Current Photovoltaic Module Recycling Processes in Europe.

Wang, T.Y., Hsiao, J.C., Du, C.H., 2012. Recycling of materials from silicon base solar cell module. Conf. Rec. IEEE Photovolt. Spec. Conf. 2355–2358.

Winter, S., Rodríguez, C., Ciroth, A., Srocka, M., 2014. GreenDelta, openLCA Advanced Training, GreenDelta. Xiu, F.R., Qi, Y., Zhang, F.

Yang, E.H., Lee, J.K., Lee, J.S., Ahn, Y.S., Kang, G.H., Cho, C.H., 2017. Environmentally friendly recovery of Ag from end-of-life c-Si solar cell using organic acid and its electrochemical purification. Hydrometallurgy 167, 129–133.

Chapter 4. Applicability and environmental benefits of alternative resource recovery technologies by substituting conventional leaching techniques

4.1. Introduction

End-of-Life (EoL) PV panels, as a new type of waste stream, have distinct characteristics. They are often installed by public sectors with large capacity (Xu et al., 2018), have a 15–20-year lifespan that could be reduced because of natural disasters (earthquakes, landslides, or hurricanes) (Mackay et al., 2014), and contain valuable materials (silver, aluminum, copper, silicon, and glass), some of which could be harmful to the environment unless treated appropriately (Jung et al., 2016; Corcelli et al., 2017). No proper PV waste management protocol has yet been established, although a few studies have focused on recycling EoL PV panels to recover raw materials (Masoumian et al., 2015; Klugmann-Radziemska and Ostrowski, 2010). Furthermore, the environmental sustainability of recycling is not ensured because of certain intensive energy and resource-consuming processes, such as acid leaching, for recovering precious metals (Latunussa et al., 2016), where nitric acid (HNO_3) leaching is applied to extract silver from cell scraps, followed by neutralization by adding calcium hydroxide ($Ca(OH)_2$).

Chemical separation of precious metals from mining ores or electronic waste involves leaching with strong solvents, such as cyanides or HNO_3 (Cui and Zhang, 2008; Coderre and Dixon, 1999), which can threaten the environment and public health by causing acidification and eutrophication (Konyratbekova et al., 2015). Recently, alternative leaching reagents were proposed as environmentally friendly and nontoxic substitutes for acids, such as thiourea (Zhang et al., 2012), thiosulfate (Petter et al., 2014), and ionic liquids (Whitehead et al., 2004; Visser et al., 2001; Dai et al., 2013). The process using such substances is termed green leaching. Cui et al. (2008) comprehensively reviewed various metallurgical resource recovery techniques from e-waste and evaluated their economic feasibility and environmental impacts. Among the hydrometallurgical alternatives, halide leaching shows potential competitiveness because of advantages such as a high leaching rate, nontoxic characteristics, reusability, and moderate reagent cost (Zhang et al., 2012). Iodide has originally been used as a replacement for cyanide for gold extraction from ore (Angelidis and Kydros, 1995; Baghalha, 2012). It has also recently been used for recovering precious metals from e-waste, such as waste printed circuit boards (PCBs) (Xiu et al., 2015). The environmental impacts of these alternatives from a life cycle perspective have not been properly addressed yet.

Several life cycle assessment (LCA) studies on PV systems have been conducted with respect to the different stages of their life cycles, i.e., (1) production/construction, (2) electricity production (i.e., operation), and (3) EoL (Vellini et al., 2017). Corcelli et al. (2018) and Ardente et al. (2019) demonstrated a comparative LCA on different recycling scenarios (i.e., high rate vs. low rate recovery) of PV panels to reflect the benefits of the recovery of valuable resources, e.g., copper, aluminum, and silver as avoided impacts. However, there is still inherent uncertainty in the database that uses existing literature, and quantitative basic unit information is needed, especially for newly developed processes.

In the previous chapter, the material recovery stage most highly contributed to the environmental impacts of the entire EoL PV treatment process. The parameters related to the material recovery process, such as the input amount of leaching agent and the recycling yields, also influenced the variance of environmental impact results higher than the parameters related to other process steps. Therefore, proper management of EoL PV panels with the recovery of valuable materials is critical, requiring environmentally sustainable solutions. This chapter proposes an application of iodine-iodide and thiourea as an alternative leaching agent for HNO_3 and evaluates the corresponding environmental impacts using experimental data. Silver and aluminum leaching from an EoL c-Si PV cell was optimized by a parametric study, and a comparative LCA is demonstrated to assess the differences in their environmental impacts.

4.2. Methodology

4.2.1. Chemical treatment and material recovery for c-Si PV waste recycling

As described in the previous chapter, material recovery in EoL PV waste treatment exhibited the highest contribution to environmental burdens. In the c-Si PV waste recycling process, chemical separation has been widely used for material recovery (Yang et al., 2017; Latunussa et al., 2016). The material recovery process was adapted by the acid leaching of ash, which remained after the thermal treatment of PV cells. During the leaching process, the ash containing metals is mixed with a solution of water and HNO_3, dissolving the metals (producing various metallic oxides) and leaving Si metal in the residues. The mixture containing the dissolved metallic oxides and Si metal residues is transferred to a vacuum filtration process, where Si metal is recovered. For recovering Si and Cu, the acid solution is successively treated by electrolysis (Latunussa et al., 2016).

4.2.2. Goal and scope

The goal of this LCA study was to compare the potential environmental impacts related to three solvents (HNO_3, iodine-iodide system (I_2–KI), and thiourea) for resource recovery from EoL c-Si PV waste. Chung et al. (2021) reported that the selective leaching of silver was accomplished by adjusting the reaction pH to 9.6, resulting in the reproducibility of 93% by rejuvenating the exhausted leaching solution. The rejuvenation of the exhausted I_2–KI solution could effectively reduce environmental impacts, especially in acidification and eutrophication, respiratory effect, and mineral extraction categories with the subsequent exclusion of the additional neutralization process.

Processing the cell scrap collected from the EoL PV panels has been defined as a function, where 2 kg (1 kg per cycle) of PV cell scraps for recovering precious metals (silver and aluminum) is defined as a FU. The system boundary in this chapter includes acid leaching, filtration, electrolysis, neutralization, and landfilling for final disposal. The environmental impact of processing 2 kg of PV cell scrap using HNO_3, I_2–KI, and thiourea solutions was compared within the system boundary.

4.2.3. Life cycle inventory of chemical treatment and material recovery

The life cycle inventory data for each process and material were derived from the Ecoinvent database. Table 4.1 shows the input and output data for the LCA.

Unlike the HNO_3 leaching method, using I_2–KI and thiourea does not require the neutralization process. In this chapter, it was assumed that the amount of Si scrap to be recovered was identical for all three leaching agents. The amount of water supplied in the leaching process for HNO_3 and thiourea and that for I_2–KI is 20 kg and 10 kg, respectively. Only a certain amount of iodine (I_2) was additionally supplied to the second cycle of the continuous process without supplying an additional water and potassium iodide (KI) input. Because the inventory of KI was unavailable in the Ecoinvent database, the inventory of potassium chloride was used in this study.

In this study, 1M HNO_3 and thiourea were used. The recovered amount of silver and aluminum was adopted from published (Chung et al., 2021) and unpublished data. The input and output data for electrolysis, neutralization, and final disposal were adopted from Latunussa et al. (2016).

Table 4.1. Input and output data of chemical treatment for c-Si PV cell scrap using different leaching agents.

	HNO_3 Leaching		I_2-KI Leaching		Thiourea Leaching	
	Amount	**Uncertainty**	**Amount**	**Uncertainty**	**Amount**	**Uncertainty**
Input						
PV cell**	2.000 kg	Lognormal dist. (σ_g, 1.0)	2.000 kg	Lognormal dist. (σ_g, 1.0)	2.000 kg	Lognormal dist. (σ_g, 1.0)
HNO_3**	2.520 kg**	Lognormal dist. (σ_g, 1.0)	—	—	—	—
I_2	—	—	1.620 kg**	Lognormal dist. (σ_g, 1.0)	—	—
KI	—	—	1.330 kg**	Lognormal dist. (σ_g, 1.0)	—	—
Thiourea	—	—	—	—	0.761 kg***	Lognormal dist. (σ_g, 1.0)
$Fe_2(SO_4)_3$	—	—	—	—	2.799 kg***	Lognormal dist. (σ_g, 1.0)
Electricity	2.580 kwh*	Lognormal dist. (σ_g, 1.2)	2.580 kwh*	Lognormal dist. (σ_g, 1.29)	2.580 Kwh*	Lognormal dist. (σ_g, 1.29)
$Ca(OH)_2$	12.992 kg*	Lognormal dist. (σ_g, 1.2)	—	—	—	—
Water	32.992 kg*	Lognormal dist. (σ_g, 1.2)	12.419 kg*	Lognormal dist. (σ_g, 1.29)	20 kg*	Lognormal dist. (σ_g, 1.29)
Output						
Limestone sludge	45.406 kg*	Lognormal dist. (σ_g, 1.2)	—	—	—	—
Inert sludge	—	—	12.504 kg#	Lognormal dist. (σ_g, 1.56)	19.905 kg#	Lognormal dist. (σ_g, 1.56)
Metal sludge	2.656 kg*	Lognormal dist. (σ_g, 1.2)	3.188 kg*	Lognormal dist. (σ_g, 1.29)	4.068 kg*	Lognormal dist. (σ_g, 1.29)
NO_x	0.712 kg*	Lognormal dist. (σ_g, 1.2)	—	—	—	—
Al recovered	0.143 kg**	Normal dist. (σ, 0.0089)	0.092 kg**	Normal dist. (σ, 0.00106)	0.00119 kg***	Normal dist. (σ, 0.00047)
Si recovered	1.576 kg*	Lognormal dist. (σ_g, 1.2)	1.576 kg*	Lognormal dist. (σ_g, 1.29)	1.576 kg*	Lognormal dist. (σ_g, 1.29)
Ag recovered	0.011 kg**	Normal dist. (σ, 0.00149)	0.008 kg**	Normal dist. (σ, 0.00017)	0.00953 kg***	Normal dist. (σ, 0.00033)

*Numerical values from Latunussa et al. (2016), **from Chung et al. (2021), *** from unpublished and # assumed data in this study

Al: aluminum; Si : silicon; Ag: silver

4.2.4. Life Cycle Impact Assessment (LCIA)

As mentioned in the previous chapter, LCA studies were conducted using OpenLCA (Winter et al., 2014) with the Ecoinvent v3.5 database. The impact assessment was performed using the ReCiPe endpoint and midpoint methods (Goedkoop et al., 2009) provided in OpenLCA (Table 4.2). In this chapter, the commonly used 11 midpoint indicators, as reported in previous studies (Wäger and Hischier, 2015; Corcelli et al., 2018), were investigated.

Table. 4.2. Life cycle impact indicators (midpoint) used in this study.

Midpoint impact category	Unit	Abbreviation
Climate change	kg CO_2-Eq	GWP
Ozone depletion	kg CFC-11-Eq	ODP
Terrestrial acidification	kg SO_2-Eq	TAP
Freshwater eutrophication	kg P-Eq	FEP
Human toxicity	kg 1,4-DCB-Eq	HTP
Photochemical oxidant formation	kg NMVOC	POFP
Terrestrial ecotoxicity	kg 1,4-DCB-Eq	TETP
Freshwater ecotoxicity	kg 1,4-DCB-Eq	FETP
Marine ecotoxicity	kg 1,4-DCB-Eq	METP
Metal depletion	kg Fe-Eq	MDP
Fossil depletion	kg oil-Eq	FDP

4.3. Results and discussion

4.3.1. Life cycle impacts of the chemical treatment process

The comparative LCA evaluates environmental impacts of chemical treatment processes using HNO_3, I_2–KI, and thiourea as leaching agents. I_2–KI and thiourea are proposed as promising alternatives in this study. Two rejuvenation cycles were considered for I_2–KI leaching. One advantage of using I_2–KI is reusability.

The results of the endpoint analysis for the chemical treatment process using three different leaching agents are presented in Fig. 4.1. These results represented only the environmental impacts, and the environmental disburdens associated with material recovery are excluded. The results indicated that the highest and lowest environmental impacts occurred when the leaching agents of HNO_3 and Thiourea, respectively, were used as leaching agents. The human health impact related to HNO_3 was found to be highest because $Ca(OH)_2$ was used for neutralization and NO_x emissions occurred during the electrolysis of HNO_3.

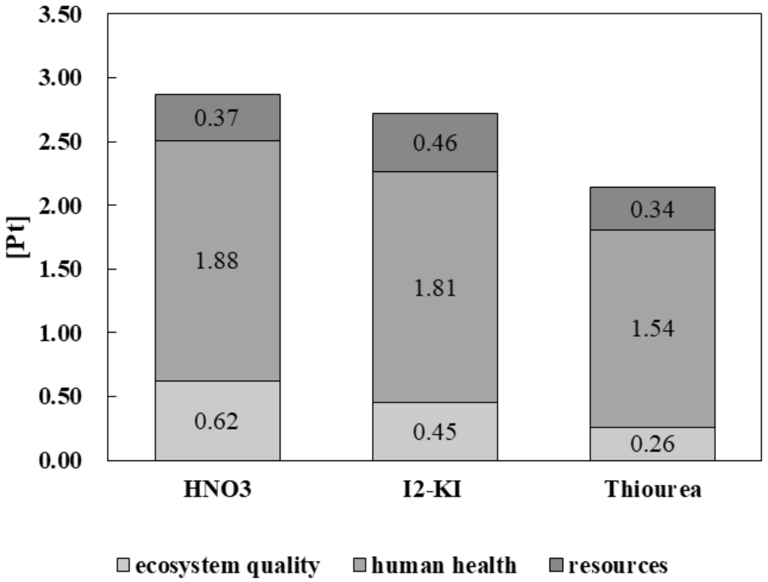


Fig. 4.1. Environmental impacts associated with the chemical treatment of 2kg PV cell scrap using three different leaching agents according to ReCiPe endpoint method.

Table 4.3 shows the environmental impact results associated with the chemical treatment of 2 kg PV cell scrap with three different leaching agents according to the ReCiPe midpoint method. The contributions of each step during the treatment are presented in Fig.4.2, with relative values to the highest impacts scaled to 100%.

For the impact categories of ODP, FEP, HTP, TETP, METP, and FDP, the chemical treatment using I_2-KI had higher environmental impacts than the other two processes. We ascribed these results to the environmental burdens related to the utilization of different leaching agents.

In the case of GWP, TAP, and POFP, the environmental impacts of the chemical treatment using HNO_3 as a leaching agent were the highest compared with the others, which was ascribed to NO_x emissions during electrolysis.

Table 4.3. In the case of GWP, TAP, and POFP, the environmental impacts of the chemical treatment using HNO_3 as a leaching agent were the highest compared with the others, which was ascribed to NO_x emissions during electrolysis.

Category	**Unit**	**HNO_3 Leaching**	**I_2-KI Leaching**	**Thiourea Leaching**
GWP	kg CO_2-Eq	1.94E+01	1.17E+01	5.36E+00
ODP	kg CFC-11-Eq	1.15E-06	1.58E-06	6.47E-07
TAP	kg SO_2-Eq	5.68E-01	4.15E-02	3.87E-02
FEP	kg P-Eq	3.07E-03	4.38E-03	3.65E-03
HTP	kg 1,4-DCB-Eq	1.14E+02	2.03E+02	1.93E+02
POFP	kg NMVOC	7.50E-01	2.74E-02	2.12E-02
TETP	kg 1,4-DCB-Eq	8.24E-03	1.23E-02	7.11E-03
FETP	kg 1,4-DCB-Eq	9.06E-02	1.74E-01	1.87E-01
METP	kg 1,4-DCB-Eq	1.01E+02	1.74E+02	1.71E+02
MDP	kg Fe-Eq	5.82E-01	3.30E-01	1.25E+00
FDP	kg oil-Eq	2.94E+00	3.74E+00	2.32E+00

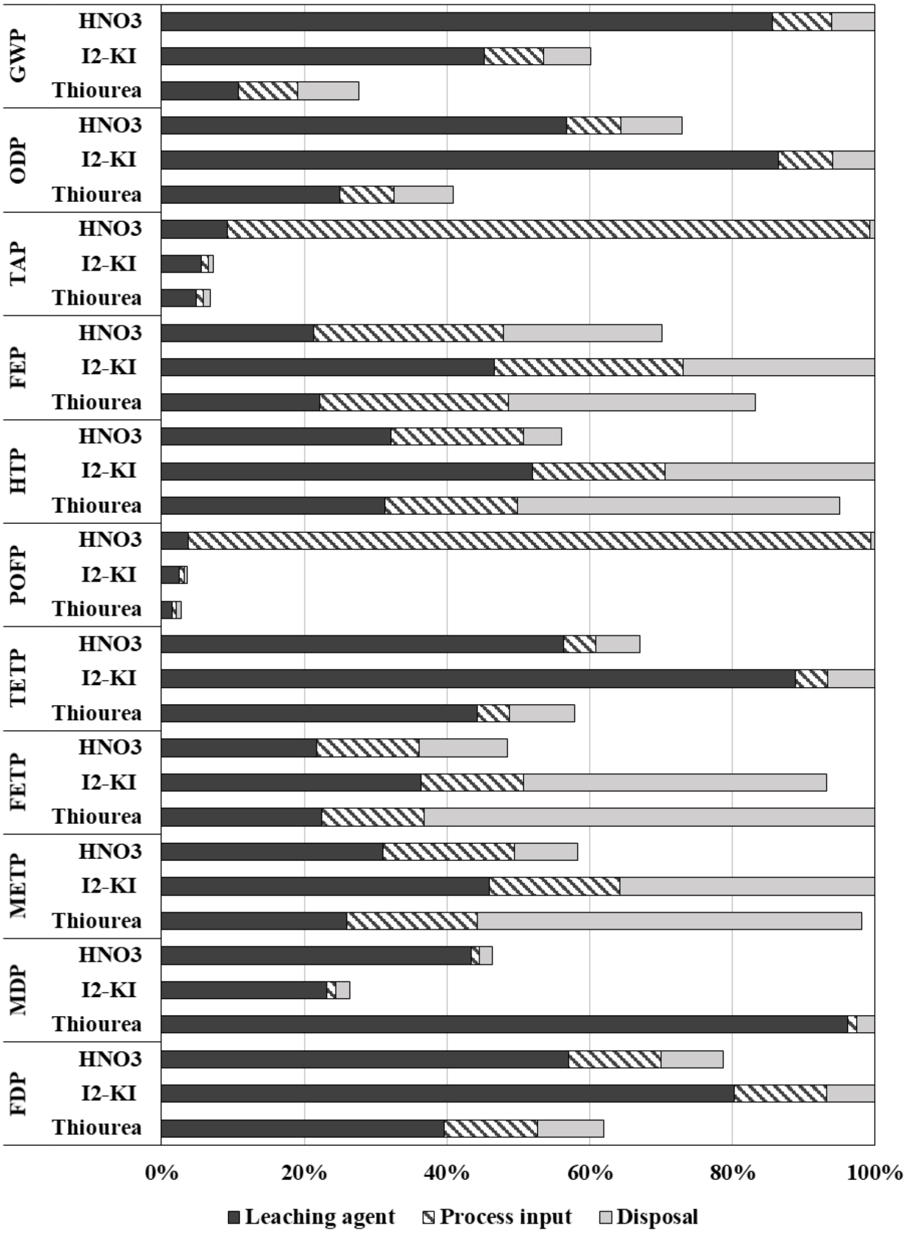


Fig. 4.2. Relative environmental impact results of three leaching agents for each midpoint indicator at individual process steps; the maximum result is set to 100%.

4.3.2. Life cycle impacts of chemical treatment considering benefits by material recovery

In this section, the system's boundaries were extended to produce secondary raw materials to quantize potential environmental benefits related to recycling, and check whether they outweigh the environmental burdens of the material recovery process.

Figure 4.3 shows the ReCiPe endpoint impacts of material recovery. From the results, the environmental benefits associated with material recovery using HNO_3 as a leaching agent were higher than those with other leaching agents. Among the recovery of aluminum, silicon, and silver, silicon recovery majorly contributed to the avoided impact.

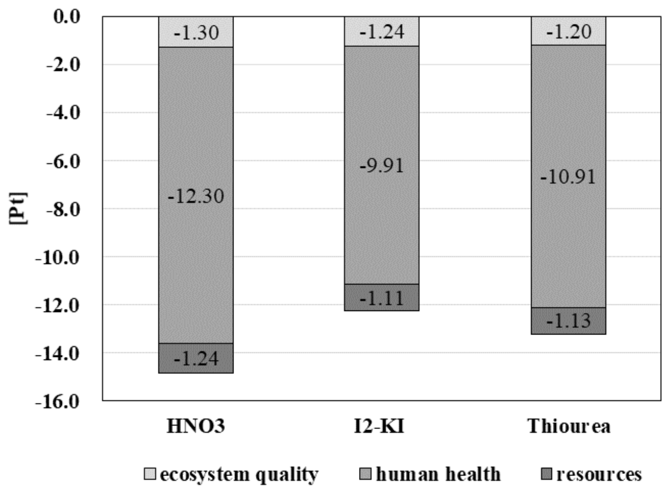


Fig. 4.3. Environmental benefits associated with material recovery of 2 kg PV cell scrap using three leaching agents using the ReCiPe endpoint method.

In Table 4.4, the overall environmental impacts associated with chemical treatment, including the impacts from the material recovery, are shown according to the endpoint method. The score (net benefit) of HNO_3 leaching was higher than the other cases. The entire environmental benefit that resulted from the material recovery was 5.2 times higher than the environmental impact by chemical treatment with HNO_3 leaching, whereas it was approximately 4.5 and 6.2 times higher in the I_2-KI and thiourea leaching processes, respectively.

Table 4.5 shows the results for the ReCiPe midpoint method. Except for ODP and TETP of the I_2-KI process and TAP and POFP of the HNO_3 process, the environmental benefits exceeded the environmental burdens by factors of 1.1 to 19.8 with HNO_3 leaching, by 1.5 to 25.3 with I_2-KI leaching, and by 1.4 to 7.6 with thiourea leaching.

Among the LCIA indicators with the three different leaching agents that were used in chemical treatment, the values of GWP, POFP, HTP, and TETP are shown in Figs. 4.4 to 4.7, and they illustrate the contribution of each process step and the recovered material to the overall environmental burdens and benefits analyzed by ReCiPe midpoint indicators.

For the ODP and TETP impact categories using I_2–KI, the detrimental impact from the entire recovery process was larger than the avoided impact. Si recovery contributed the highest to the impacts avoided via material recovery. For the HTP, FETP, METP, FEP, and MDP impact categories, the avoided impacts related to material recovery were remarkably higher than the environmental burdens. For these impact categories, the environmental disburdenment associated with silver recovery accounted for more than 77% of the total avoided impact.

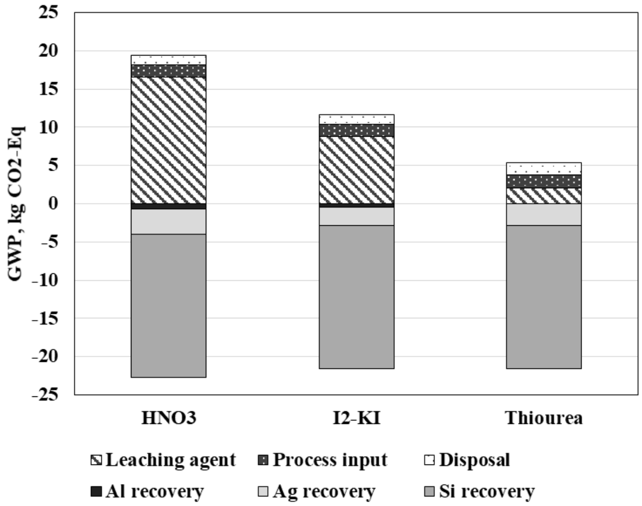


Fig. 4.4. GWP results responsible for three different leaching agents exhibiting contribution to each treatment and recovery process.

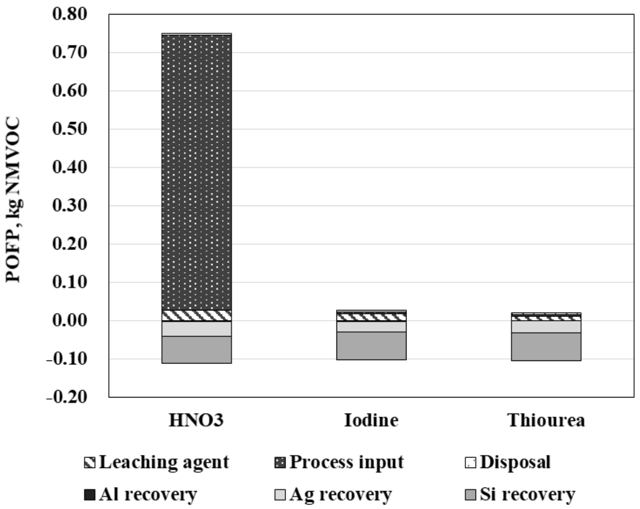


Fig. 4.5. POFP results responsible for three different leaching agents exhibiting contribution to each treatment and recovery process.

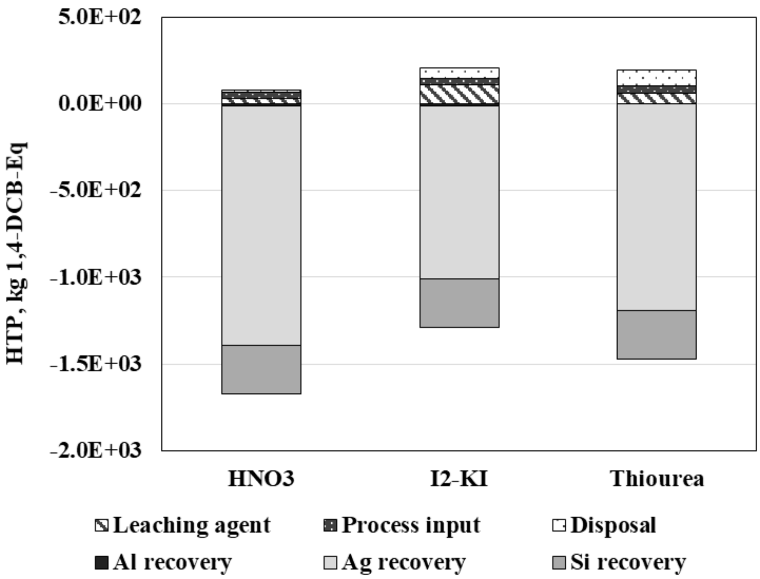


Fig. 4.6. HTP results responsible for three different leaching agents exhibiting contribution to each treatment and recovery process.

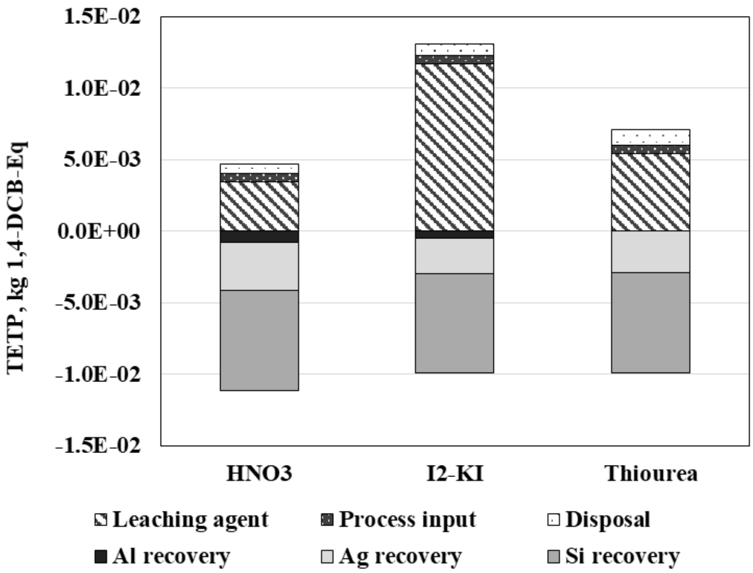


Fig. 4.7. TETP results responsible for three different leaching agents exhibiting contribution to each treatment and recovery process.

Table 4.4. Total environmental impacts of chemical treatment and material recovery of 2 kg PV cell scraps with three leaching agents using the ReCiPe midpoint method.

Impact category	Unit	HNO_3 leaching			I_2-KI leaching			Thiourea leaching		
		Impact of chemical treatment	Impact of material recovery	Total impact	Impact of chemical treatment	Impact of material recovery	Total impact	Impact of chemical treatment	Impact of material recovery	Total impact
Ecosystem quality	**Pt**	0.62	−1.30	−0.67	0.45	−1.24	−0.79	0.26	−1.20	−0.93
Human health	**Pt**	1.88	−12.30	−10.42	1.81	−9.91	−8.10	1.54	−10.91	−9.36
Resources	**Pt**	0.37	−1.24	−0.88	0.46	−1.11	−0.64	0.34	−1.13	−0.80
Total impacts	**Pt**	2.87	−14.84	−11.97	2.72	−12.25	−9.53	2.14	−13.23	−11.10

Table 4.5. Total environmental impacts of chemical treatment and material recovery of 2-kg PV cell scraps with three leaching agents using the ReCiPe midpoint method.

Impact category (Unit)	**Leaching method**	**Leaching agent**	**Process input**	**Disposal**	**Al recovery**	**Ag recovery**	**Si recovery**	**Total**
GWP ($kg\ CO_2$-Eq)	HNO_3	1.66E+01	1.63E+00	1.18E+00	−6.93E-01	−3.35E+00	−1.87E+01	−3.36E+00
	I_2–KI	8.76E+00	1.63E+00	1.26E+00	−4.46E-01	−2.44E+00	−1.87E+01	−9.94E+00
	Thiourea	2.08E+00	1.63E+00	1.65E+00	−5.74E-03	−2.90E+00	−1.87E+01	−1.63E+01
ODP (kg CFC-11-Eq)	HNO_3	9.00E-07	1.20E-07	1.35E-07	−5.80E-08	−2.81E-07	−8.78E-07	−6.22E-08
	I_2–KI	1.37E-06	1.20E-07	9.39E-08	−3.73E-08	−2.04E-07	−8.78E-07	4.64E-07
	Thiourea	3.96E-07	1.20E-07	1.31E-07	−4.81E-10	−2.43E-07	−8.78E-07	−4.74E-07
TAP (kg SO2-Eq)	HNO_3	5.22E-02	5.11E-01	4.64E-03	−3.99E-03	−4.12E-02	−9.67E-02	4.26E-01
	I_2–KI	3.20E-02	5.43E-03	4.07E-03	−2.56E-03	−3.00E-02	−9.67E-02	−8.77E-02
	Thiourea	2.77E-02	5.43E-03	5.52E-03	−3.30E-05	−3.57E-02	−9.67E-02	−9.38E-02
FEP (kg P-Eq)	HNO_3	9.36E-04	1.16E-03	9.74E-04	−2.70E-04	−1.81E-02	−8.62E-03	−2.39E-02
	I_2–KI	2.04E-03	1.16E-03	1.18E-03	−1.74E-04	−1.32E-02	−8.62E-03	−1.76E-02
	Thiourea	9.70E-04	1.16E-03	1.52E-03	−2.24E-06	−1.57E-02	−8.62E-03	−2.07E-02
HTP (kg 1,4-DCB-Eq)	HNO_3	6.53E+01	3.77E+01	1.10E+01	−1.37E+01	−1.38E+03	−2.78E+02	−1.56E+03
	I_2–KI	1.06E+02	3.77E+01	5.99E+01	−8.83E+00	−1.00E+03	−2.78E+02	−1.09E+03
	Thiourea	6.38E+01	3.77E+01	9.18E+01	−1.14E-01	−1.19E+03	−2.78E+02	−1.28E+03
POFP	HNO_3	2.86E-02	7.17E-01	4.85E-03	−2.41E-03	−3.72E-02	−7.21E-02	6.38E-01

(kg NMVOC)	I_2–KI	1.88E-02	4.53E-03	4.03E-03	−1.55E-03	−2.71E-02	−7.21E-02	−7.34E-02
	Thiourea	1.11E-02	4.53E-03	5.52E-03	−2.00E-05	−3.23E-02	−7.21E-02	−8.32E-02
TETP (kg 1,4-DCB-Eq)	HNO_3	6.91E-03	5.59E-04	7.66E-04	−7.59E-04	−3.37E-03	−7.00E-03	−2.88E-03
	I_2–KI	1.09E-02	5.56E-04	8.19E-04	−4.88E-04	−2.45E-03	−7.00E-03	2.35E-03
	Thiourea	5.44E-03	5.57E-04	1.11E-03	−6.29E-06	−2.92E-03	−7.00E-03	−2.81E-03
FETP (kg 1,4-DCB-Eq)	HNO_3	4.06E-02	2.70E-02	2.30E-02	−3.64E-01	−5.96E-01	−2.03E-01	−1.07E+00
	I_2–KI	6.79E-02	2.69E-02	7.95E-02	−2.34E-01	−4.34E-01	−2.03E-01	−6.96E-01
	Thiourea	4.19E-02	2.70E-02	1.18E-01	−3.02E-03	−5.17E-01	−2.03E-01	−5.35E-01
METP (kg 1,4-DCB-Eq)	HNO_3	5.41E+01	3.19E+01	1.54E+01	−1.04E+02	−8.74E+02	−2.44E+02	−1.12E+03
	I_2–KI	7.98E+01	3.19E+01	6.22E+01	−6.67E+01	−6.36E+02	−2.44E+02	−7.73E+02
	Thiourea	4.51E+01	3.19E+01	9.37E+01	−8.59E-01	−7.58E+02	−2.44E+02	−8.32E+02
MDP (kg Fe-Eq)	HNO_3	5.43E-01	1.55E-02	2.27E-02	−3.11E-02	−1.13E+01	−1.56E-01	−1.09E+01
	I_2–KI	2.91E-01	1.54E-02	2.39E-02	−2.00E-02	−8.22E+00	−1.56E-01	−8.07E+00
	Thiourea	1.20E+00	1.54E-02	3.31E-02	−2.58E-04	−9.79E+00	−1.56E-01	−8.70E+00
FDP (kg oil-Eq)	HNO_3	2.13E+00	4.84E-01	3.26E-01	−1.71E-01	−1.04E+00	−4.71E+00	−2.98E+00
	I_2–KI	3.00E+00	4.84E-01	2.54E-01	−1.10E-01	−7.53E-01	−4.71E+00	−1.84E+00
	Thiourea	1.48E+00	4.84E-01	3.50E-01	−1.42E-03	−8.97E-01	−4.71E+00	−3.29E+00

Al: aluminum; Si: silicon; Ag: silver

4.3.3. Uncertainty analysis of chemical treatment

In this section, an uncertainty analysis was performed related to the environmental impacts for the chemical treatment of c-Si PV cell scraps using three leaching agents. Uncertainty analyses were conducted using Monte Carlo simulation with 1000 runs, calculating the probability of the results based on random sampling (Harrison et al., 2010). In this chapter, the estimated data adapted from the previous literature (Latunussa et al., 2016; Chung et al., 2019) were assumed to have a lognormal distribution with geometric standard deviations, σ_g, which were chosen using the Pedigree matrix (Weidema and Wesnaes, 1996). For the input data related to HNO_3 leaching, 1.20 σ_g was used, whereas 1.29 σ_g and 1.56 σ_g were chosen for the material consumed and the amount of disposal, respectively, for I_2–KI and thiourea leaching. However, the recovered amount was assumed to have a normal distribution with standard deviations adapted from the experimental result. Table 4.1 shows the standard deviations for each material.

Figure 4.8 shows the results of uncertainty analysis, revealing that the uncertainty values were lower than the differences in environmental impacts for each chemical treatment method. However, the uncertainty values of 5% and 95% percentile for the material recovery were within the differences in the environmental impacts between each result (Fig. 4.9). For the statistical analysis, the results were obtained by performing the ANOVA tests. The results revealed that the impact variances from different leaching agents showed meaningful differences ($p < 0.05$) (Tables 4.6 and 4.7).

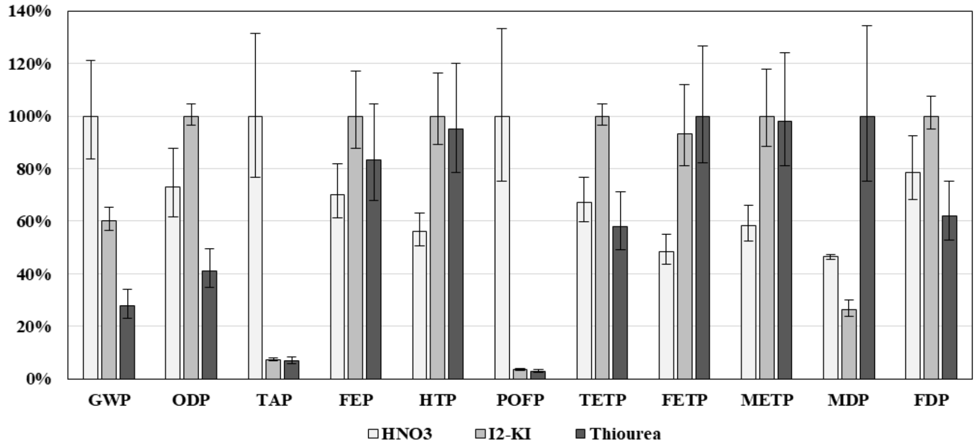


Fig. 4.8. Environmental impacts comparison of chemical treatment using three leaching agents with uncertainty analysis. The error bar indicates 90% confidence intervals.

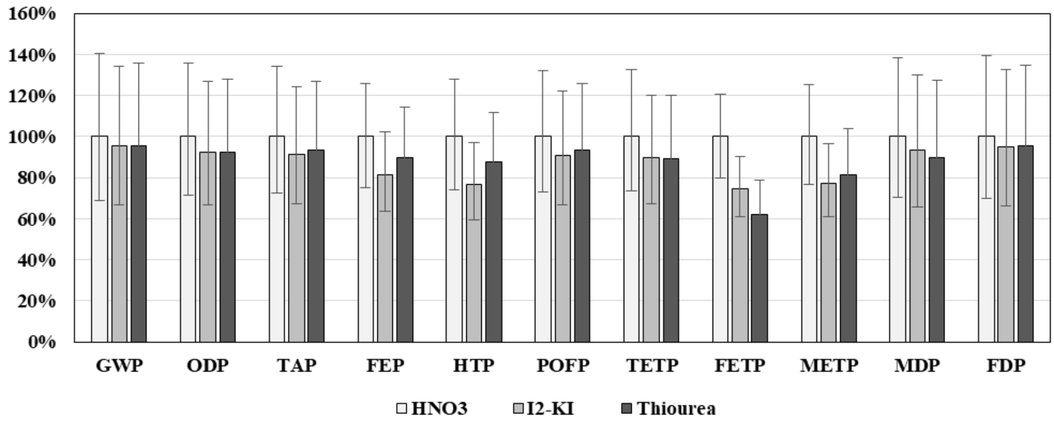


Fig. 4.9. Environmental impacts comparison of material recovery using three leaching agents with uncertainty analysis. The error bar indicates 90% confidence intervals.

Table 4.6. Uncertainty analysis of environmental impacts related to chemical treatment using three leaching agents with ANOVA test results.

Impact Category	unit	HNO_3			I_2-KI			Thiourea			F-Value	p-Value
		Mean	5% Percentile	95% Percentile	Mean	5% Percentile	95% Percentile	Mean	5% Percentile	95% Percentile		
GWP	kg CO2-Eq	1.94E+01	1.62E+01	2.35E+01	1.17E+01	1.10E+01	1.27E+01	5.36E+00	4.49E+00	6.60E+00	1919.9	0.000
ODP	kg CFC-11-Eq	1.15E-06	9.75E-07	1.39E-06	1.58E-06	1.53E-06	1.67E-06	6.47E-07	5.54E-07	7.81E-07	9792.6	0.000
TAP	kg SO2-Eq	5.68E-01	4.36E-01	7.46E-01	4.15E-02	3.93E-02	4.50E-02	3.87E-02	3.28E-02	4.82E-02	1867.5	0.000
FEP	kg P-Eq	3.07E-03	2.68E-03	3.58E-03	4.38E-03	3.83E-03	5.28E-03	3.65E-03	2.97E-03	4.59E-03	558.0	0.000
MEP	kg N-Eq	1.14E+02	1.03E+02	1.28E+02	2.03E+02	1.75E+02	2.60E+02	1.93E+02	1.60E+02	2.44E+02	2822.2	0.000
HTP	kg 1,4-DCB-Eq	7.50E-01	5.65E-01	1.00E+00	2.74E-02	2.54E-02	3.06E-02	2.12E-02	1.82E-02	2.56E-02	2472.6	0.000
POFP	kg NMVOC	8.24E-03	7.33E-03	9.43E-03	1.23E-02	1.20E-02	1.28E-02	7.11E-03	6.05E-03	8.74E-03	1169.9	0.000
TETP	kg 1,4-DCB-Eq	9.06E-02	8.14E-02	1.03E-01	1.74E-01	1.44E-01	2.35E-01	1.87E-01	1.54E-01	2.37E-01	6229.4	0.000
FETP	kg 1,4-DCB-Eq	1.01E+02	9.13E+01	1.15E+02	1.74E+02	1.48E+02	2.27E+02	1.71E+02	1.41E+02	2.16E+02	2660.0	0.000
MDP	kg Fe-Eq	5.82E-01	5.72E-01	5.95E-01	3.30E-01	3.21E-01	3.45E-01	1.25E+00	9.44E-01	1.68E+00	1736.5	0.000
FDP	kg oil-Eq	2.94E+00	2.55E+00	3.46E+00	3.74E+00	3.55E+00	4.04E+00	2.32E+00	1.97E+00	2.82E+00	3350.9	0.000

Table 4.7. Uncertainty analysis of environmental benefits related to material recovery using three leaching agents with ANOVA test results.

Impact Category	unit	HNO_3			I_2-KI			Thiourea			F-Value	p-Value
		Mean	5% Percentile	95% Percentile	Mean	5% Percentile	95% Percentile	Mean	5% Percentile	95% Percentile		
GWP	kg CO_2-Eq	−2.3E+01	−3.3E+01	−1.6E+01	−2.2E+01	−3.1E+01	−1.5E+01	−2.2E+01	−3.1E+01	−1.5E+01	8.55	0.000
ODP	kg CFC-11-Eq	−1.2E-06	−1.7E-06	−8.9E-07	−1.1E-06	−1.6E-06	−8.3E-07	−1.1E-06	−1.6E-06	−8.2E-07	12.53	0.000
TAP	kg SO_2-Eq	−1.4E-01	−1.9E-01	−1.0E-01	−1.3E-01	−1.8E-01	−9.7E-02	−1.4E-01	−1.8E-01	−9.9E-02	6.79	0.001
FEP	kg P-Eq	−2.8E-02	−3.5E-02	−2.1E-02	−2.2E-02	−2.8E-02	−1.7E-02	−2.5E-02	−3.1E-02	−1.9E-02	70.67	0.000
MEP	kg N-Eq	−1.7E+03	−2.2E+03	−1.3E+03	−1.3E+03	−1.7E+03	−1.0E+03	−1.5E+03	−1.9E+03	−1.1E+03	209.23	0.000
HTP	kg 1,4-DCB-Eq	−1.1E-01	−1.5E-01	−8.3E-02	−1.0E-01	−1.4E-01	−7.6E-02	−1.1E-01	−1.4E-01	−7.8E-02	10.97	0.000
POFP	kg NMVOC	−1.1E-02	−1.5E-02	−8.4E-03	−1.0E-02	−1.4E-02	−7.6E-03	−1.0E-02	−1.4E-02	−7.4E-03	18.32	0.000
TETP	kg 1,4-DCB-Eq	−1.2E+00	−1.4E+00	−9.5E-01	−8.8E-01	−1.1E+00	−7.2E-01	−7.3E-01	−9.3E-01	−5.7E-01	969.84	0.000
FETP	kg 1,4-DCB-Eq	−1.2E+03	−1.6E+03	−9.5E+02	−9.6E+02	−1.2E+03	−7.6E+02	−1.0E+03	−1.3E+03	−7.8E+02	251.76	0.000
MDP	kg Fe-Eq	−7.5E-04	−1.0E-03	−5.3E-04	−7.0E-04	−9.8E-04	−5.0E-04	−6.8E-04	−9.6E-04	−4.6E-04	361.58	0.000
FDP	kg oil-Eq	−6.0E+00	−8.4E+00	−4.2E+00	−5.7E+00	−8.0E+00	−4.0E+00	−5.8E+00	−8.1E+00	−4.0E+00	4.72	0.010

4.4. Summary

The goal of this LCA was to compare the potential environmental impacts related to three different leaching agents (i.e., HNO_3, I_2-KI, and thiourea) for material recovery from EoL c-Si PV waste. The system boundary in this chapter included acid leaching, filtration, electrolysis, neutralization and landfilling for final disposal. The environmental impacts produced from 2 kg PV cell scrap using HNO_3, I_2-KI, and thiourea solution were compared within the system boundary.

The ReCiPe results revealed that the highest environmental impacts were produced from chemical treatment using HNO_3 as a leaching agent and the lowest was produced by thiourea leaching for the endpoint levels. The human health impact related to HNO_3 leaching was the highest because of using $Ca(OH)_2$ for neutralization and NO_x emissions during HNO_3 electrolysis.

For the ReCiPe midpoint method, the environmental impacts associated with HNO_3 leaching were the highest for the TAP, PMFP, MEP, and POFP impact categories, whereas those related to I_2–KI leaching were the highest for GWP, ODP, TETP, and FDP. The environmental impacts produced from thiourea leaching were the highest for FETP and MDP.

The overall environmental impacts associated with chemical treatment, including the impacts from material recovery, revealed that the score (net benefit) of HNO_3 leaching was higher than that of the other two cases because of more recovered materials, such as aluminum, silver, and silicon.

The uncertainty analysis of the environmental impacts associated with the chemical treatment and material recovery process was performed using Monte-Carlo simulation with 1,000 runs. The impact variances from different leaching agents showed meaningful statistical differences determined by the ANOVA test (p-Value < 0.05).

A part of this chapter has been published as a research paper in Journal of Hazardous Materials (404, 123989), 2021.

References

Angelidis, T.N., Kydros, K.A., 1995. Selective gold dissolution from a roasted auriferous pyrite-arsenopyrite concentrate. Hydrometallurgy 37, 75–88.

Ardente, F., Latunussa, C.E.L., Blengini, G.A., 2019. Resource efficient recovery of critical and precious metals from waste silicon PV panel recycling. Waste Manag. 91, 156–167.

Baghalha, M., 2012. The leaching kinetics of an oxide gold ore with iodide/iodine solutions. Hydrometallurgy 113–114, 42–50.

Chung, J., Seo, B., Lee, J., Kim, J.Y., 2021. Comparative analysis of I2-KI and HNO3 leaching in a life cycle perspective: Towards sustainable recycling of end-of-life c-Si PV panel. J. Hazard. Mater. 404, 123989.

Coderre, F., Dixon, D.G., 1999. Modeling the cyanide heap leaching of cupriferous gold ores. Part 1: Introduction and interpretation of laboratory column leaching data. Hydrometallurgy 52, 151–175.

Corcelli, F., Ripa, M., Leccisi, E., Cigolotti, V., Fiandra, V., Graditi, G., Sannino, L., Tammaro, M., Ulgiati, S., 2018. Sustainable urban electricity supply chain – Indicators of material recovery and energy savings from crystalline silicon photovoltaic panels end-of-life. Ecol. Indic. 94, 37–51.

Cui, J., Zhang, L., 2008. Metallurgical recovery of metals from electronic waste: A review. J. Hazard. Mater. 158, 228–256.

Dai, Y., van Spronsen, J., Witkamp, G.J., Verpoorte, R., Choi, Y.H., 2013. Natural deep eutectic solvents as new potential media for green technology. Anal. Chim. Acta 766, 61–68.

Goedkoop, M., Heijungs, R., Huijbregts, M., Schryver, A. De, Struijs, J., Zelm, R. Van, 2009. ReCiPe 2008 a life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. Report I: Characterisation.

Harrison, R.L., Granja, C., Leroy, C., 2010. Introduction to Monte Carlo Simulation, in: AIP Conference Proceedings. IEEE, pp. 17–21.

Jung, B., Park, J., Seo, D., Park, N., 2016. Sustainable System for Raw-Metal Recovery from Crystalline Silicon Solar Panels: From Noble-Metal Extraction to Lead Removal. ACS Sustain. Chem. Eng. 4, 4079–4083.

Klugmann-Radziemska, E., Ostrowski, P., 2010. Chemical treatment of crystalline silicon solar cells as a method of recovering pure silicon from photovoltaic modules. Renew. Energy 35, 1751–1759.

Konyratbekova, S.S., Baikonurova, A., Akcil, A., 2015. Non-cyanide leaching processes in gold hydrometallurgy and iodine-iodide applications: A review. Miner. Process. Extr. Metall. Rev. 36, 198–212.

Latunussa, C.E.L., Ardente, F., Blengini, G.A., Mancini, L., 2016. Life Cycle Assessment of an innovative recycling process for crystalline silicon photovoltaic c. Sol. Energy Mater. Sol. Cells 156, 101–111.

Mackay, A.A., Gan, P., Yu, R., Smets, B.F., 2014. Seasonal arsenic accumulation in stream sediments at a groundwater discharge zone. Environ. Sci. Technol. 48, 920–929.

Masoumian, M., Kopacek, P., 2015. End-of-Life Management of Photovoltaic Modules. IFAC-PapersOnLine 48, 162–167.

Petter, P.M.H., Veit, H.M., Bernardes, A.M., 2014. Evaluation of gold and silver leaching from printed circuit board of cellphones. Waste Manag. 34, 475–482.

Vellini, M., Gambini, M., Prattella, V., 2017. Environmental impacts of PV technology throughout the life cycle: Importance of the end-of-life management for Si-panels and CdTe-panels. Energy 138, 1099–1111.

Visser, A.E., Swatloski, R.P., Griffin, S.T., Hartman, D.H., Rogers, R.D., 2001. Liquid/liquid extraction of metal ions in room temperature ionic liquids. Sep. Sci. Technol. 36, 785–804.

Wäger, P.A., Hischier, R., 2015. Life cycle assessment of post-consumer plastics production from waste electrical and electronic equipment (WEEE) treatment residues in a Central European plastics recycling plant. Sci. Total Environ. 529, 158–167.

Weidema, B.P., Wesnæs, M.S., 1996. Data quality management for life cycle inventories—an example of using data quality indicators. J. Clean. Prod. 4, 167–174.

Whitehead, J.A., Lawrance, G.A., McCluskey, A., 2004. “Green” leaching: Recyclable and selective leaching of gold-bearing ore in an ionic liquid. Green Chem.

6, 313–315.

Winter, S., Rodríguez, C., Ciroth, A., Srocka, M., GreenDelta, 2014. openLCA Advanced Training. GreenDelta.

Xiu, F.R., Qi, Y., Zhang, F.S., 2015. Leaching of Au, Ag, and Pd from waste printed circuit boards of mobile phone by iodide lixiviant after supercritical water pre-treatment. Waste Manag. 41, 134–141.

Xu, Y., Li, J., Tan, Q., Peters, A.L., Yang, C., 2018. Global status of recycling waste solar panels: A review. Waste Manag. 75, 450–458.

Yang, E.H., Lee, J.K., Lee, J.S., Ahn, Y.S., Kang, G.H., Cho, C.H., 2017. Environmentally friendly recovery of Ag from end-of-life c-Si solar cell using organic acid and its electrochemical purification. Hydrometallurgy 167, 129–133.

Zhang, Y., Liu, S., Xie, H., Zeng, X., Li, J., 2012. Current Status on Leaching Precious Metals from Waste Printed Circuit Boards. Procedia Environ. Sci. 16, 560–568.

Chapter 5. Environmental impacts analysis under a high-level EoL treatment scenario focusing on PV waste transportation in the Republic of Korea

5.1. Introduction

While the recycling of end-of-life (EoL) c-Si PV waste can reduce environmental burdens because the secondary production of material requires less energy, produces less pollution, and has a lower cost than the extraction of primary materials, efficient collection and transportation also are essential components to ensure that the recycling system is economically and environmentally beneficial. Xu and his co-workers (2013) reported that transportation is an important factor that affects the results of the life cycle assessment for the management of waste glass from cathode ray tube funnels. According to a previous study in which a life cycle assessment (LCA) was performed for the collection, transportation, treatment, and disposal of municipal waste (De Feo et al., 2016), the global warming category was affected significantly by transportation due to the lack of nearby facilities.

PV waste can create several economic and environmental burdens because the materials in PV cells are expensive and highly toxic (Fthenakis, 2000). In addition, the long-distance transportation of the EoL PV panels to the recycling site causes adverse environmental impacts, which can be considered as a crucial barrier to the PV recycling system (Deng et al., 2019). According to Latunussa et al. (2016), the transportation of EoL PV panels to the recycling site contributed mainly to climate change, accounting for 30% of the total contribution among all of the impact categories that were evaluated. On the contrary, Held (2013) reported that transportation during the recycling process imposed fewer environmental burdens. These different outcomes can be ascribed to the assumptions regarding transportation varying under different scenarios, and the distance measure for recycling differed between each case.

To date, only a few studies have been conducted concerning the impact of the transportation of PV wastes. In this chapter, the environmental impacts associated with the transportation of EoL PV panels are evaluated on the basis of the scenarios for recycling that have been proposed in the Republic of Korea. To achieve this goal, transportation scenarios were developed based on two different scenarios, and a comparative analysis also was performed that reflected the actual domestic conditions. Therefore, the novelty of this work is that it uses actual travel distance data in the Republic of Korea to assess the environmental burdens associated with the transportation of EoL PV panels from each discharge point to landfills under different conditions.

5.2. Methodology

5.2.1. Description of process for transporting PV waste

As shown in Fig. 5.1., the PV waste recycling process includes (1) transporting the EoL PV panels from the origin to the collection center, (2) performing pretreatment for recycling, and (3) landfilling residues. However, the number of recycling facilities is often limited because of high capital costs.

Fig. 5.1. General scheme of the process used to recycle the EoL PV panels from their origin to the landfill.

This chapter describes the design and use of the two scenarios shown in Fig. 5.2. Transport A excludes transportation to the collection center, meaning that the EoL PV panels are directly transported to the recycling facility for reclamation. The residues unfeasible for recycling are transported to a landfill for their final disposal. In Transport B, EoL PV planes are transported to the regional collection center, where the aluminum frame and junction box are disassembled during pretreatment. The disassembled PV wastes are transported to the recycling facility. After recycling, other processes are the same as those of Transport A.

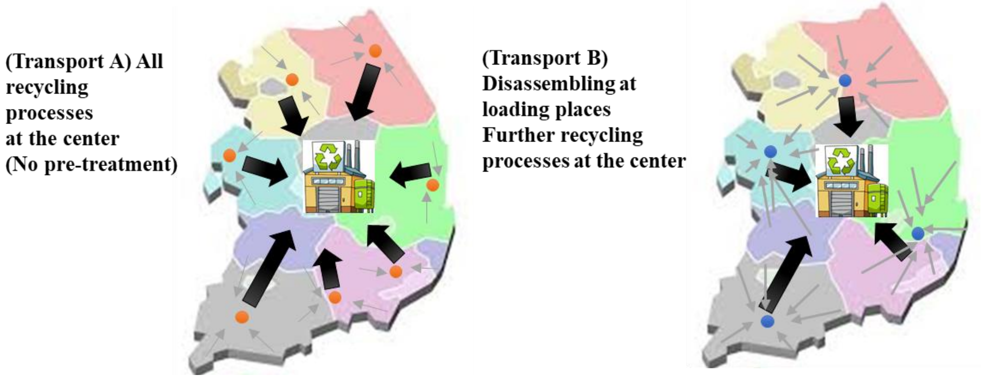


Fig. 5.2. Two scenarios for EoL PV panels: (a) directly transported to the recycling facility (Transport A), and (b) via four collection centers with disassembling before recycling PV panels (Transport B).

5.2.2. Identification of the location of the recycling facility, collection centers, and final disposal site

In Korea, the construction of the recycling facility for EoL PV panels is underway according to the Base construction project for solar recycling center construction (Kim et al., 2019; Chungcheongbuk-do, 2017). In this study, the transportation distances were calculated based on the location information of the recycling site, as shown in Table 5.1.

Table 5.1. Information of the Base construction project for solar recycling center construction (Chungcheongbuk-do, 2017).

Project	**Base construction project for solar recycling center construction**
Location	Euntan-ri, Munbaek-myeon, Jincheon-gun, Chungcheongbuk-do
Participants	Chungbuk Technopark, Korea Institute of Energy Research, etc.
Area	15,935 m^2

The Korea Ministry of Environment has established and announced the construction project for collection centers in the metropolitan area, Yeongnam, Chungcheong, and Honam regions, as shown in Table 5.2. Thus, the locations of the four collection centers were used for estimations of all distances in Transport B.

Table 5.2. Information of the Base construction project for collection center of future resource recycle.

Zone	Location	Construction scale
Metropolitan area	91, Gongdan 2-daero, Siheung-si, Gyeonggi-do	1,480 m^2
Yeongnam region	40, Seongseogongdan-ro, Dalseo-gu, Daegu	2,053 m^2
Chungcheong region	228, Geumma-ro, Geumma-myeon, Hongseong-gun, Chungcheongnam-do	900 m^2
Honam region	849-2, Habuk-dong, Jeongeup-si, Jeollabuk-do	1,700 m^2

Residues after the recycling process were assumed to be disposed of finally in landfills because high-level EoL treatment in Chapter 3 was proposed as the recycling scenario that was practically applied in Korea. The landfill candidates were selected from the National Waste Statistical Survey (ME, 2018) by considering the end date of landfilling after 2040 and open landfill for industrial waste, as shown in Table 5.3.

Table 5.3. Information of landfills for final disposal selected in this study.

Name	Address	Close year (Year)
Gyeonggi Ecoland	39, Cheonghak-ro 8beon-gil, Byeollae-myeon	2050
Gwangju Sanitary Landfill	160, Dodong-gil, Nam-gu	2075
Jeonbuk Waste Landfill	45, Iseong-ri, Iseo-myeon, Wanju-Gun	2043
Gyeongbuk Waste Landfill	499, Songbaek-ro, Jangcheon-myeon, Gumi-si	2065

5.2.3. Specification of the direct transport scenario without a collection center (Transport A)

As shown in Fig. 5.3, the two most important parameters to analyze the environmental impacts by transportation are the distance traveled and the amount transported. This scenario consisted of route 1 and 2.

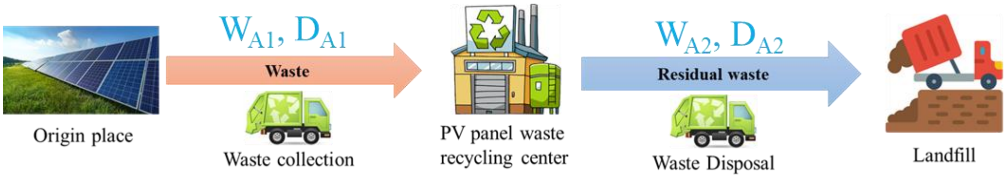


Fig. 5.3. Transport process and parameters used in Transport A.

1) Estimated amount of PV wastes (W_{A1})

Based on the annual installation capacity of PV panels according to the dissemination statistics of renewable energy from 2004 to 2017 with an assumption of 100 tons of PV waste generated per MW (KEI, 2018), the amount of PV wastes in the years from 2029 to 2043 was forecasted using the following equation (Paiano, 2015):

$W_y = \sum_{x=1}^{y} u_x W$, where:

u_x = MW/year

W = weight (t) per MW

x = year

y = year of waste generation (x + 25)

In the above approach, the assumption was made that the yearly installed weight amount of PV panels would be equal to the weight of the PV wastes generated 25 years later.

2) Distance from origins to the recycling facility (D_{A1})

The transportation distances from the discharge points where EoL PV panels are discarded to the recycling facility were estimated based on the following assumptions. . The PV wastes initially are expected to be collected at 242 public screening facilities for recyclable resources, which were assumed to be the origin because these facilities are closely distributed near possible discharge points in Korea (ME, 2018).

Based on these assumptions, the transport distance was calculated as follows. (1) Calculate the average distance from public screening facilities in 17 regions to the recycling facility and (2) calculate the weighted average distance (D_{A1}) by estimating each amount of PV waste generated in every 17 regions. The transportation distances were estimated by measuring the distances between each 242 public screening facilities to the recycling facility using the geographic distribution on Google maps, considering distance and travel time. Particularly, PV waste generated in Jeju Island is transported by ships additionally; therefore, the amount of PV waste transported by ship (W_j) and the shipping distance between Moko port and Jeju port (D_j).

In this study, it was assumed that all of the PV wastes would be treated at the recycling facility, and the reuse of the PV wastes was not considered. This decision was made because the efficiency would be lowered as time passed if PV wastes were reused and the aluminum frame and junction box were replaced (Lunardi et al., 2018).

3) Estimated amount of residue transported to the landfill after recycling (W_{A2})

The final amount of waste to be sent to the landfill was based on the high-level EoL scenarios in Chapter 3. The amount of residue after the recycling process of 1 ton of PV waste was 562.5 kg, which was composed of 13.2 kg of glass waste, 3.1 kg of fly ash from the thermal treatment, and 469.2 kg and 77.0 kg of liquid waste and sludge waste, respectively, produced from chemical material separation.

4) Distance from the recycling facility to the landfill (D_{A2})

This distance from the recycling facility to the landfill was calculated using the geographic distribution on Google maps, considering distance and travel time.

5.2.4. Specification of the transportation scenario via collection center (Transport B)

In Transport B, the transport distances from the discharge points to the collection centers, from the collection centers to the recycling facility, and from the recycling facility to landfill were defined as route 1, 2, and 3, respectively, as depicted in Fig. 5.4.

Fig. 5.4. Transport process and parameters used in Transport B.

1) Estimated amount of PV wastes (W_{B1})

W_{B1} used in Transport B was identical to that of Transport A.

2) Distance from the origin to the collection center (D_{B1})

The transport distance, D1, was calculated based on the average distance from 242 public screening facilities in 17 regions to each of the four collection centers that have been identified. Then, the weighted average distance (D_{B1}) was calculated on the basis of estimating the amounts of PV wastes generated in all 17 regions.

3) Estimated amount of PV waste after pre-treatment (W_{B2})

In Transport B, the aluminum frame and junction box were assumed to be disassembled from the EoL PV panels at each collection center. By adopting the weight of the junction box of 10 kg and aluminum frame of 159.5 kg disassembled per 1 ton of PV waste in Chapter 3, W2 was estimated at 830.5 kg of pretreated PV waste per 1 ton of EoL PV, which was transported to the recycling facility.

4) Distance from the collection center to the recycling facility (D_{B2})

The distance was calculated as the weighted average distance from each of the four collection centers to the recycling facility after pre-treatment based on the estimated PV waste amounts generated at the four regions.

5) Estimated amount of residue after recycling (W_{B3}) and distance from the recycling facility to the landfill (D_{B3})

The values of W_{B3} and D_{B3} were identical to W_{A2} and D_{A2} used in Transport A, respectively.

5.2.5. Life cycle assessment

This study was focused on the effect of transportation on the environmental impacts of EoL treatment processes of PV waste. The functional unit was 1 ton of c-Si PV wastes. The system boundary was set as the entire transport process from the origins to the landfills.

With respect to transportation, the lifecycle inventory datasets used in this study are presented in Table 5.4. The impact assessment was performed by means of the midpoint ReCiPe methods provided by OpenLCA (Winter et al., 2014) with the Ecoinvent v3.5 database. Six of the 18 ReCiPe midpoint categories were found in Chapter 3 to be highly influenced by transportation in Chapter 3, were included in this study, and they were global warming potential (GWP, in kg CO_2-Eq), ozone depletion (ODP, in kg CFC-11-Eq), human toxicity (HTP, in kg 1,4-DCB-Eq), terrestrial ecotoxicity (TETP, in kg 1,4-DCB-Eq), metal depletion (MDP, in kg Fe-Eq), and fossil depletion (FDP, in kg oil-Eq).

Table 5.4. Datasets used in this study for transportation of 1 ton PV waste.

Transport step	Input parameter (kg × km)	Dataset
Transport A- route 1 Transport B- route 1	$1000 \times D_{A1}$ $1000 \times D_{B1}$	transport, freight, lorry3.5–7.5 metric ton, EURO6, RoW
Transport B- route 2	$830.5 \times D_{B2}$	transport, freight, lorry7.5–16 metric ton, EURO6, RoW
Transport A- route 2 Transport B- route 3	$562.5 \times D_{A2}$ $562.5 \times D_{A3}$	
Transport by ship	$Wj \times Dj$	transport, freight, sea, transoceanic ship, GLO

5.3. Results and discussion

5.3.1. Input parameters of the transportation scenarios

Based on the yearly installed PV capacity from 2004 to 2018, the projected amounts of PV waste from 2029 to 2043 are shown in Table 5.5. In 2029, the PV waste amount is forecasted at 255 tons, and it increases by almost a factor of 100 in 2033. The projected PV waste amount abruptly increases after 2037 and is projected at 236,720 tons in 2043, which is 1000 times higher than that in 2029. From Fig. 5.5, the distributed discharged PV waste amounts in 17 regions differ annually. In broad outlines, however, the generated PV wastes from the Jeonnam, Gyeongbuk, and Jeonbuk regions account for a major portion, except in 2029.

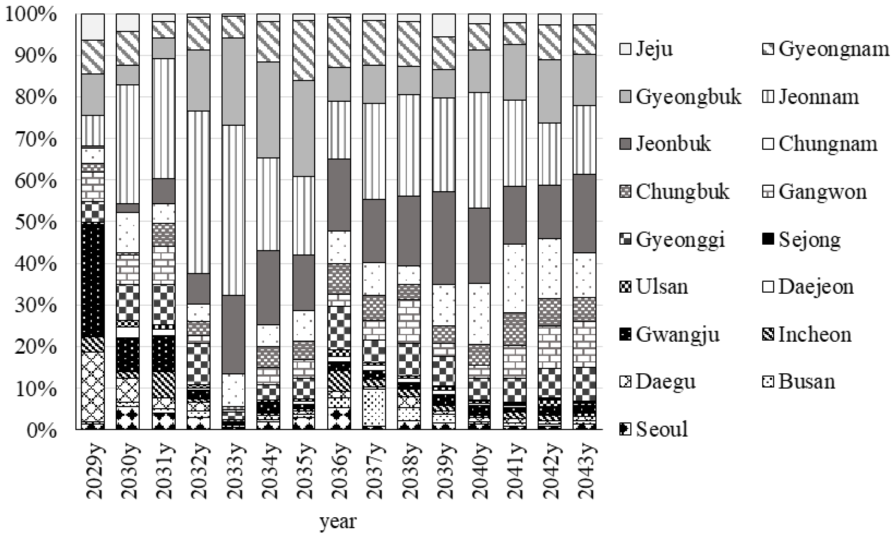


Fig. 5.5. Regional distribution of PV waste projected from 2029 to 2043 in the Republic of Korea.

Table 5.5. Estimated distance (D_{A1}) of Transport A based on the amount of PV waste generated (ton/year).

	Calculated distance		Estimated amount of PV waste(ton/year)														
Region	No. of Points	Average Distance(km)	2029y	2030y	2031y	2032y	2033y	2034y	2035y	2036y	2037y	2038y	2039y	2040y	2041y	2042y	2043y
Seoul	17	118.7	4	28	92	133	161	338	360	420	296	1,152	1,429	1,455	839	1,110	3,046
Busan	16	306.0	1	5	24	74	132	91	105	178	2,574	1,670	1,945	809	585	720	2,428
Daegu	8	193.1	43	29	54	92	88	158	97	134	162	1,418	927	964	996	1,111	1,888
Incheon	22	167.2	9	9	140	41	55	97	79	392	576	918	1,198	850	1,359	1,681	2,084
Gwangju	5	228.0	69	41	193	91	70	369	132	157	565	920	2,446	2,640	1,113	2,498	3,777
Daejeon	5	56.1	0	13	35	22	58	70	100	116	398	504	948	477	376	659	871
Ulsan	5	284.8	2	8	27	37	100	87	76	124	176	263	561	510	125	1,812	986
Sejong	1	27.2	-	-	-	-	-	-	-	-	-	65	335	366	556	852	1,020
Gyeonggi	35	110.2	13	43	216	452	488	597	636	819	1,653	4,129	6,622	5,887	5,254	9,794	19,208
Gangwon	19	200.6	18	35	202	85	179	696	560	224	1,390	5,538	2,937	3,707	7,196	13,906	26,199
Chungbuk	10	61.4	5	1	126	154	200	820	563	585	1,734	2,012	3,704	5,463	7,229	8,800	13,591
Chungnam	14	98.1	9	49	106	196	2,193	895	912	608	2,315	2,306	9,185	16,657	14,897	19,509	25,462
Jeonbuk	13	154.9	2	10	134	325	5,152	2,973	1,712	1,364	4,469	8,914	20,831	20,530	12,557	17,510	45,054
Jeonnam	22	274.5	19	143	638	1,777	11,319	3,689	2,369	1,099	6,823	12,967	20,896	31,701	19,020	20,609	38,927
Gyeongbuk	22	177.4	25	23	116	657	5,783	3,840	2,921	634	2,692	3,516	6,273	11,352	11,977	20,726	29,027
Gyeongnam	23	248.3	21	42	87	361	1,416	1,623	1,827	961	3,176	5,818	7,090	7,113	4,947	11,282	16,839
Jeju*	5	147.3	16	21	43	40	175	315	216	67	517	961	5,300	2,911	1,898	3,669	6,314
W_{A1} (ton)			255	499	2,232	4,535	27,567	16,684	12,665	7,882	29,516	53,072	92,626	113,390	90,922	136,249	236,720
D_{A1} (km)		167.9	197.7	198.2	193.7	207.5	208.9	190.3	188.7	174.8	198.3	197.0	183.4	185.1	173.2	174.0	176.0

*Additional distance of transport by shipping: 178 km

The travel distance from the origin to the recycling facility (D_{A1}) in Transport A was calculated as the weighted average of the distances from 242 origins to recycling centers with the projected PV waste amount for 17 regions. As shown in Table 5.5, the arithmetical mean of travel distances of 17 regions was calculated to be 167.9 km. Considering the annually estimated PV waste amount, it differed from 173.2 to 208.9 km as the average values of 189.8 km. Additional transportation for PV waste generated in Jeju region was considered. (1) The shipping travel distance was estimated to be 178 km and (2) the PV waste amount was estimated to be 6.3 kg to 63.1 kg per total waste amount as 1 ton.

The travel distances from the recycling facilities to the landfills (D_{A2}) of Transport A was estimated to be 124.6 km, 243.0 km, 147.5 km, and 149.2 km for each landfill selected in this study, as listed in Table 5.6. In the subsequent analysis, the default value was set to 125.6 km for Landfill A, which was the shortest distance.

Table 5.6. Estimated the distance from the recycling facility to the landfill.

Landfill option	Landfill	D_{A2}, D_{B3} (km)
Landfill A	Gyeonggi Ecoland	124.6
Landfill B	Gwangju Sanitary Landfill	243.0
Landfill C	Jeonbuk Waste Landfill	147.5
Landfill D	Gyeongbuk Waste Landfill	149.2

Table 5.7. Summarized input parameters of Transport A.

Parameters	Input amount	Data range
D_{A1} (km)	189.8	173.2 – 208.9
D_{A2} (km)	124.6	147.5, 149.2, 243.0
D_j (km)	178.0	-
W_j (kg)	25.1	6.3 – 63.1

The travel distances from the origins to the collection centers (D_{B1}) in Transport B were calculated as the weighted average of the distances from 242 origins to four collection centers, which managed the PV waste generated at each origin. The projected PV waste amounts at the 17 regions were used as the weighting factor. As shown in Table 5.8, the arithmetical mean of the travel distance in the 17 regions was calculated to be 93.4 km. Considering the annually estimated PV waste amount, its weighted average value was 93.4 km, which differed from 81.7 km to 101.4 km. Additional transportation for PV waste generated in the Jeju region was considered the same as D_j for Transport A.

While predicting that 830.5 kg of W_{B2} was produced from 1 ton of W_{B1}, Table 5.9 shows the weighted average travel distance for W_{B2} from the four collection centers to the recycling facility. The arithmetical mean of the travel distance from the four collection centers (D_{B2}) was 144.3 km. Considering the disassembled amount of PV wastes year by year, the weighted distance varied from 147.1 to 167.6 km.

The travel distance from the recycling facility to the landfill (D_{B3}) in Transport B was estimated to be the same as D_{A2} in Transport A .

Table 5.8. Estimated distance (D_{B1}) of Transport B based on the amount of PV waste generated (ton/year).

Region	Estimated distance		Estimated amount of PV waste (ton/year)														
	No. of Points	Average Distance(km)	2029y	2030y	2031y	2032y	2033y	2034y	2035y	2036y	2037y	2038y	2039y	2040y	2041y	2042y	2043y
Seoul	17	46.9	4	28	92	133	161	338	360	420	296	1,152	1,429	1,455	839	1,110	3,046
Busan	16	122.1	1	5	24	74	132	91	105	178	2,574	1,670	1,945	809	585	720	2,428
Daegu	8	12	43	29	54	92	88	158	97	134	162	1,418	927	964	996	1,111	1,888
Incheon	22	58.2	9	9	140	41	55	97	79	392	576	918	1,198	850	1,359	1,681	2,084
Gwangju	5	61.8	69	41	193	91	70	369	132	157	565	920	2,446	2,640	1,113	2,498	3,777
Daejeon	5	94.5	0	13	35	22	58	70	100	116	398	504	948	477	376	659	871
Ulsan	5	130.8	2	8	27	37	100	87	76	124	176	263	561	510	125	1,812	986
Sejong	1	77.5	-	-	-	-	-	-	-	-	-	65	335	366	556	852	1,020
Gyeonggi	35	59.6	13	43	216	452	488	597	636	819	1,653	4,129	6,622	5,887	5,254	9,794	19,208
Gangwon	19	209.3	18	35	202	85	179	696	560	224	1,390	5,538	2,937	3,707	7,196	13,906	26,199
Chungbuk	10	138.9	5	1	126	154	200	820	563	585	1,734	2,012	3,704	5,463	7,229	8,800	13,591
Chungnam	14	54.7	9	49	106	196	2,193	895	912	608	2,315	2,306	9,185	16,657	14,897	19,509	25,462
Jeonbuk	13	58.1	2	10	134	325	5,152	2,973	1,712	1,364	4,469	8,914	20,831	20,530	12,557	17,510	45,054
Jeonnam	22	115	19	143	638	1,777	11,319	3,689	2,369	1,099	6,823	12,967	20,896	31,701	19,020	20,609	38,927
Gyeongbuk	22	92.4	25	23	116	657	5,783	3,840	2,921	634	2,692	3,516	6,273	11,352	11,977	20,726	29,027
Gyeongnam	23	106.4	21	42	87	361	1,416	1,623	1,827	961	3,176	5,818	7,090	7,113	4,947	11,282	16,839
Jeju*	5	149.2	16	21	43	40	175	315	216	67	517	961	5,300	2,911	1,898	3,669	6,314
W_{B1} (ton)			255	499	2,232	4,535	27,567	16,684	12,665	7,882	29,516	53,072	92,626	113,390	90,922	136,249	236,720
D_{B1} (km)		93.4	81.7	94.2	98.9	96.2	93.5	95.3	95.9	87.5	99.1	101.4	91.7	91.8	97.2	99.9	99.4

*Additional distance of transport by shipping: 178 km

Table 5.9 Estimated distance (D_{B2}) of Transport B based on the amount of PV waste generated (ton/year).

Region	Estimated Distance (km)	Estimated amount of PV waste (ton/year)														
		2029y	2030y	2031y	2032y	2033y	2034y	2035y	2036y	2037y	2038y	2039y	2040y	2041y	2042y	2043y
Metropolitan	122.2	35	94	526	575	715	1,400	1,324	1,503	3,171	9,507	9,870	9,637	11,864	21,458	40,935
Chungcheong	92.2	12	52	217	301	1,985	1,445	1,276	1,060	3,602	3,959	11,479	18,600	18,677	24,154	33,164
Honam	169.7	85	173	817	1,809	13,540	5,950	3,587	2,176	10,022	19,247	40,073	46,803	28,016	35,871	76,199
Yeongnam	192.9	74	85	248	989	6,089	4,697	4,070	1,645	7,112	10,276	13,605	16,806	15,090	28,878	41,446
W_{B2}(ton)		207	404	1,808	3,673	22,329	13,492	10,258	6,384	23,908	42,988	75,027	91,846	73,647	110,362	191,743
D_{B2}(km)	144.3	165.4	153.7	149.8	162.2	167.6	164.5	163.1	151.6	158.6	157.6	155.8	153.3	147.1	149.6	151.2

Table 5.10. Summarized input parameters of Transport B.

Parameters	**Input amount**	**Data range**
D_{B1} (km)	93.4	81.7–101.4
D_{B2} (km)	144.3	147.1–167.6
D_{B3} (km)	124.6	147.5, 149.2, 243.0
D_j (km)	178.0	—
W_j (kg)	25.1	6.3–63.1

5.3.2. Impacts of EoL PV panels for transportation scenarios

The calculated environmental impacts for the two transportation scenarios are shown in Fig. 5.6, in which relative values to the high impacts are scaled to 100%. The environmental impacts of Transport B exhibited in the range of 73% to 86%, which were lower than those of Transport A. The sum of the travel distance in Transport B was 362.3 km, which was longer than that in Transport A (313.4 km). Nevertheless, the lower environmental impacts in Transport B were obtained because the aluminum frame and the junction box were removed during the pre-treatment of the PV wastes at the collection centers. Therefore, the amount of PV wastes to be transported in Transport B was reduced compared with Transport A.

Furthermore, fuel consumption and emissions were reduced when PV waste was transported by a large-sized vehicle after being collected at the collection center. Therefore, the environmental efficacy of transportation increases when using a large-sized vehicle.

The error bars represent the differences produced from the variations of the travel distance, which were calculated by the weighted average based on the amount of PV wastes generated at each region, which is used as a weighting factor.

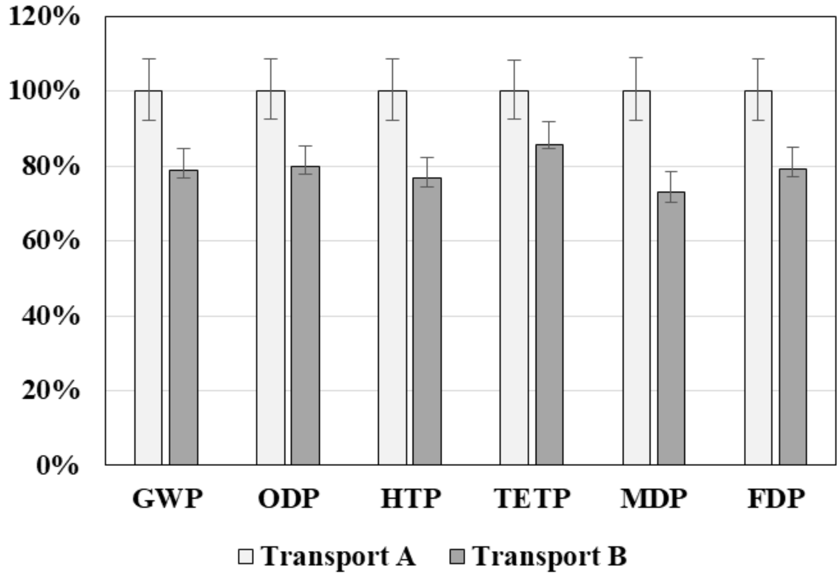


Fig. 5.6. Relative environmental impacts under the two different scenarios for each ReCiPe indicator; the higher result is set to 100%. The error bars indicate the difference caused by the variations of the input parameters.

The contribution of each transport to the entire EoL treatment is presented in Fig. 5.7. Here, the system boundary was expanded to the entire EoL treatment process. With respect to the GWP, the impact by route 1 and 2 in Transport A was 109.1 kg CO_2-eq, indicating that it contributed 19.9% to the entire EoL treatment. Transport A exhibited that the impact caused by routet 1 was approximately 6 to 8 times higher than that by route 2. In Transport B, the GWP impact caused by the entire transport was calculated as 86.0 kg CO_2-eq. This contribution to the entire EoL treatment was found to be 15.7%. With respect to the environmental indicators of ODP, TETP, and MDP, the contribution percentage related to transportation in Transport A was calculated to be more than 60%. However, the contribution percentage that resulted from the indicator of TETP in Transport B was found to be 58.9%, which was the most significant among the six different impact categories.

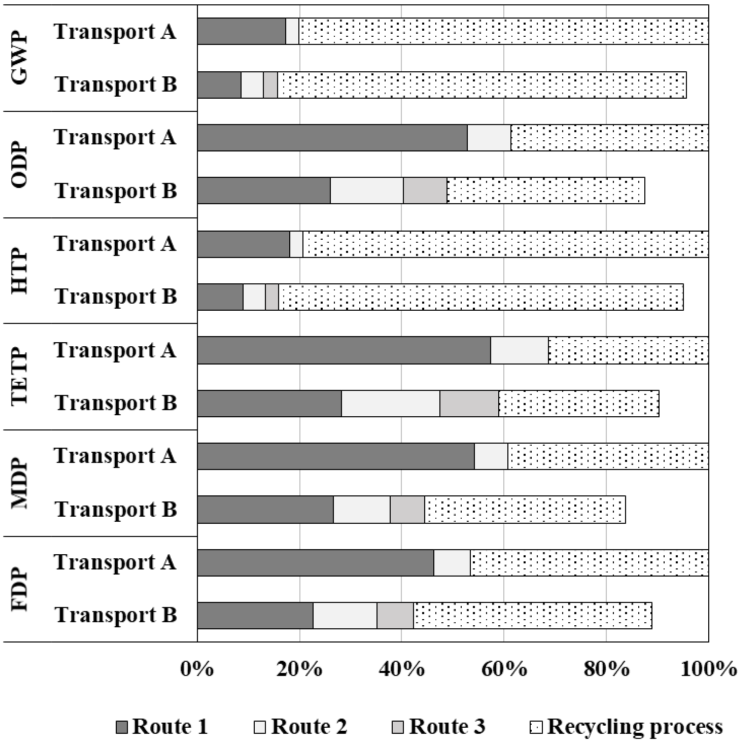


Fig. 5.7. Relative environmental impacts of the two different scenarios for each ReCiPe indicator by each process; the higher result is set to 100%; the system boundary was expanded to the entire EoL treatment process.

In this chapter, four landfill candidates were selected for the final disposal of PV waste. Figure 5.8 shows the relative environmental impacts of the cases with the travel distance to each landfill, D_{A2} and D_{B3} in Transports A and B, respectively. As the travel distance to the landfill increased from 124.6 km to 243.0 km, the environmental impact of MDP and TETP in Transport A increased to 9.9% and 15.1%, respectively. In Transport B, however, those values were calculated to be 14.6% and 18.5%, respectively.

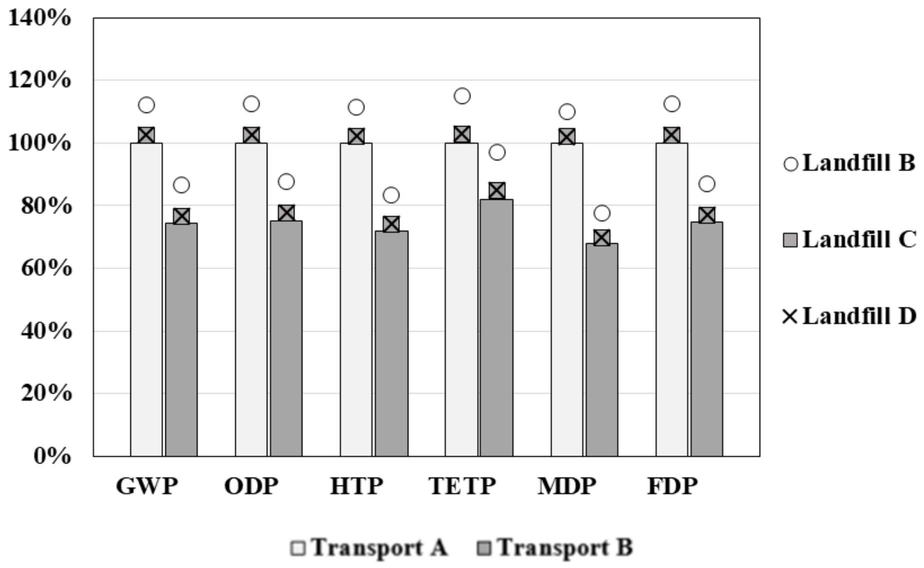


Fig. 5.8. Relative environmental impacts for the four different transport scenarios, including the effect of the variance in travel distances to the selected landfills.

5.3.3. Impacts of the transportation of EoL PV panels: vehicle size and type

In this study, it was assumed that the vehicles used for transportation were 5- and 10-ton lorry based on the inventory of "lorry 3.5–7.5 t" and "lorry 7.5–16 t" in the Ecoinvent database. As PV waste is expected to increase rapidly from 2034, it is necessary to use a large-capacity vehicle for efficient transportation. Lunardi et al. (2018) proposed that "lorry 16–32 t" would be used for transporting PV waste to the recycling facility. Also, Latunussa et al. (2016) have assumed that different types of vehicles are used at each transport stage: for example, lorry 3.5–7.5 t for transporting PV waste to a local collection point, lorry 16–32 t for transporting PV waste to a recycling plant, and lorry 3.5–7.5 t for transporting residuals to a landfill.

In this section, a new scenario was set up as shown in Table 5.11. Transport A-1 was composed of route 1 by “lorry 7.5–16 t” from the origin to the recycling facility and route 2 by “lorry 16–32 t” from the recycling facility to the landfill. Other input parameters were the PV waste amount and travel distances for the transportation used in Transport A. Transport B-1 included Transport B by “lorry 16–32 t” from the collection centers to the recycling facility with a FU of 1 ton of PV waste.

Table 5.11. Options with different sizes of vehicles for transportation used in this chapter.

	Transport A-1	**Transport B-1**
Route 1	7.5 – 16 t lorry	7.5 – 16 t lorry
Route 2	16 – 32 t lorry	16 – 32 t lorry
Route 3	-	16 – 32 t lorry

As a result of the analysis, when a large vehicle was used, the environmental impacts in Transport A-1 with respect to all indicators except TETP were reduced by more than 50% as compared to the existing scenario (Transport A) as shown in Fig. 5.9. In Transport B-1, those were reduced by more than 40% as compared to Transport B.

When comparing Transport A-1 and Transport B-1, the environmental impacts of Transport B-1 were found to be still lower than Transport A-1, but the difference was reduced to 1.5 ~ 8.5%. In particular, with respect to TETP, the impact of Transport B-1 was found to be higher than that of Transport B-1. This is mainly resulted from the increase of travel distance while the transport via the collection center does not offset the effect of the reduced amount of PV waste. In addition, the energy and material consumption as well as the emissions were reduced when a large-sized vehicle was used, but the decreases of these material flows between “lorry 3.5-7.5t” and “lorry 7.5-16t” were found to be smaller than those between “lorry 7.5-16t” and “lorry 16-32t”.

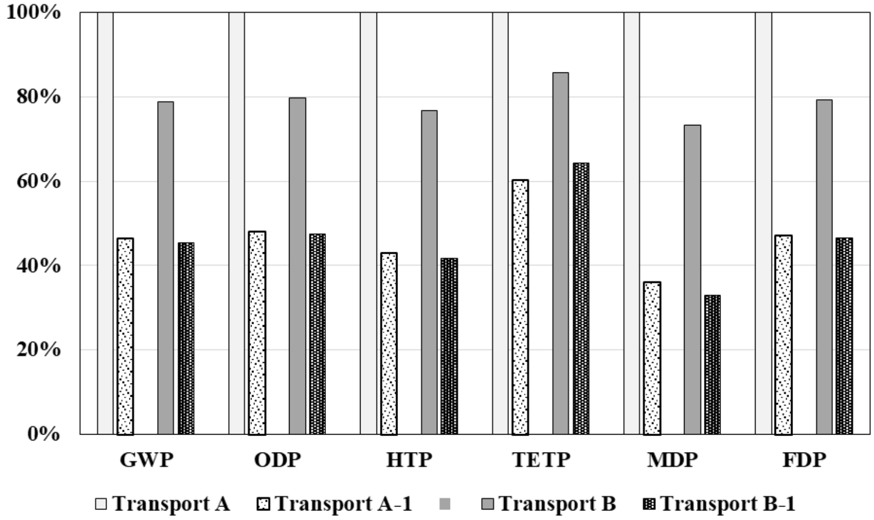


Fig. 5.9. Relative environmental impacts with two vehicle size options for each scenario; the higher result of Transport A is set to 100%.

In the transportation sector, the issues related to increasing energy efficiency and decreasing emissions has boomed up the use of electric vehicles as future transport (Bachman, et al., 2015). Electric vehicles (EVs) have been considered as a promising technology to reduce green-house gas emissions compared to internal combustion engine vehicles. Nevertheless, there are only a few research studies focused on whether the benefits can offset the environmental impacts on the supply of electricity required for charging and associated with production of battery. Previous studies reported that their environmental benefits were highly dependent on the energy mix that provides electricity to charge EVs and the lifetime of EVs and battery (Petrauskiene et al., 2020; Kawamoto et al., 2019; Giradi et al., 2015).

EVs are mainly used as passenger cars because of their efficiency, but the freight movement with EVs has been recently implemented in some countries. Therefore, this study analyzed the difference in environmental impacts when transporting PV wastes by trucks that use electricity instead of diesel fuel. It has been assumed that the vehicle size, travel distance, and the amount of PV wastes in this case are identical to the baseline scenario used in the previous session. Since the inventory of electric truck was not available in the Ecoinvent database, the input flows have been calculated based on the inventory data for an electric passenger car applying the ratio between the fuel consumption of diesel trucks and passenger car. The electric consumption of lorry 3.5-7.5t per one-ton kilometer (tkm) was calculated to be 0.398 kWh/tkm, while that of lorry 7.5-16t was 0.169 kWh/tkm.

Fig. 5.10 shows the relative environmental impacts from the transport of PV wastes associated with the options for different vehicle energy sources. In the case of GWP, ODP, and FDP, the environmental impacts of transportation options by EVs (Transport A-2 and B-2) were reduced by 23% to 60% compared to those obtained by diesel vehicles. These results were mainly resulted from the emissions related to fuel consumption and the production of fuels for diesel vehicles which decreased due to replacement with EVs. For the impact categories of HTP, TETP, and MDP, on the other hand, the environmental impacts of the transportation by EVs increased up to 103% to 514% which were much higher than those of diesel vehicles used in this study. The increase of HTP is mainly due to the production of lithium-ion battery. And, other impacts, i.e. TETP, and MDP, related to the electricity production varied with the energy mix such as technologies and fuel types (Petrauskiene, et al., 2020).

The values of the environmental impacts in each scenario shown in Table 5.12.

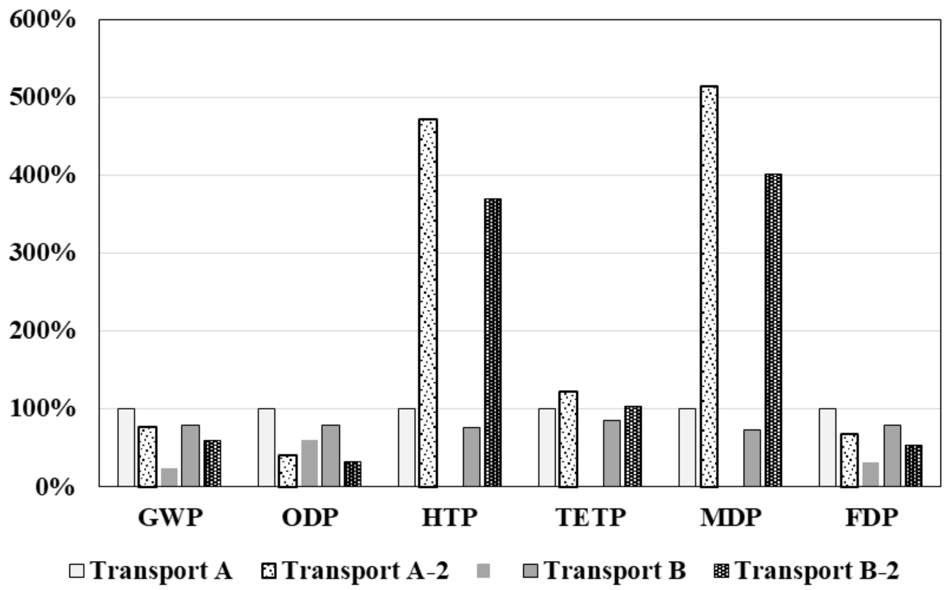


Fig. 5.10. Relative environmental impacts dependent on the type of vehicle (diesel vehicles in Transports A and B; EVs in Transports A-2 an B-2); the result of Transport A is set to 100%.

Table 5.12. The environmental impacts of 1-ton EoL PV transportation under different scenarios using the ReCiPe midpoint method.

Impact category	**Unit**	**Baseline Scenario**		**Scenario of Large-sized vehicle**		**Scenario of electricity vehicle**	
		Transport A	Transport B	Transport A-1	Transport B-1	Transport A-2	Transport B-2
GWP	kg CO2-Eq	1.09E+02	8.61E+01	5.07E+01	4.96E+01	8.37E+01	6.56E+01
ODP	kg CFC-11-Eq	1.94E-05	1.54E-05	9.32E-06	9.22E-06	7.77E-06	6.18E-06
HTP	kg 1,4-DCB-Eq	8.78E+02	6.74E+02	3.77E+02	3.65E+02	4.14E+03	3.25E+03
TETP	kg 1,4-DCB-Eq	4.46E-01	3.82E-01	2.69E-01	2.86E-01	5.45E-01	4.60E-01
MDP	kg Fe-Eq	5.69E+00	4.16E+00	2.06E+00	1.88E+00	2.93E+01	2.28E+01
FDP	kg oil-Eq	4.15E+01	3.29E+01	1.96E+01	1.93E+01	2.83E+01	2.22E+01

5.4. Summary

In this chapter, the environmental burdens were quantified focusing on transportation during EoL PV treatment. Two scenarios were developed under the assumptions of employing one recycling facility and four regional collection centers where a proper pretreatment process for EoL PV treatment should be directed in Korea. It was assumed that the PV waste amount was projected based on one ton of PV waste generated per 10 kW electricity production with a lifespan of 25 years. The weighted distance from the origin to the collection and recycling centers was calculated using the weighting factor as the amount of regional PV waste, including shipping transport from Jeju Island. The four candidate landfills for final disposal were selected considering residual landfill capacity, remaining periods, and permission for industrial waste.

In Transport A, the travel distance was calculated to be 313.4 km, shorter than that of Transport B with pretreatment at the collection center, which was 362.3 km. The ReCiPe midpoint results revealed that the environmental impacts were reduced in the range of −26.8% to −14.2% in Transport B compared with Transport A. As the travel distance to the landfill increased from 124.6 km to 243.0 km, the environmental impacts in Transport A increased to 9.9% and 15.1%, whereas those values increased to 14.6% and 18.5% in Transport B.

Scenarios A-1 and B-1, with a large-sized vehicle, were developed to evaluate the effect of the vehicle size associated with transportation. When a large-sized vehicle was used, the environmental impacts for all indicators, except TETP, reduced by more than 50% in Transport A-1. However, the environmental impacts when using a large-sized vehicle in Transport B-1 reduced by less than 8.5%.

For using EVs as future transport, the inventory for electric trucks was modified to compare the environmental impacts depending on the vehicle types. For the impact categories of GWP, ODP, and FDP, the environmental impacts associated with EVs reduced by 23%–60%. However, for HTP, TETP, and MDP, the environmental impacts with the transportation using EVs increased by 103%–514%, which is much higher than those of diesel vehicles.

References

Bachmann, C., Chingcuanco, F., MacLean, H., Roorda, M.J., 2015. Life-cycle assessment of diesel-electric hybrid and conventional diesel trucks for deliveries. J. Transp. Eng. 141, 1–8.

Chung-cheong bukdo Press, 2017, “Confirmation of site for base construction project for solar power recycling center construction,”

De Feo, G., Ferrara, C., Iuliano, C., Grosso, A., 2016. LCA of the collection, transportation, treatment and disposal of source separated municipal waste: A Southern Italy case study. Sustain. 8.

Deng, R., Chang, N.L., Ouyang, Z., Chong, C.M., 2019. A techno-economic review of silicon photovoltaic module recycling. Renew. Sustain. Energy Rev. 109, 532–550.

Fthenakis, V.M., 2000. End-of-life management and recycling of PV modules. Energy Policy 28, 1051–1058.

Girardi, P., Gargiulo, A., & Brambilla, P. C. (2015). A comparative LCA of an electric vehicle and an internal combustion engine vehicle using the appropriate power mix: the Italian case study. International Journal of Life Cycle Assessment, 20(8), 1127–1142.

Held, M. (2013) LCA screening of a recycling process for silicon based PV modules. PV Cycle Conf. 2013, 1–19.

Kawamoto, R., Mochizuki, H., Moriguchi, Y., Nakano, T., Motohashi, M., Sakai, Y., & Inaba, A. (2019). Estimation of CO2 Emissions of Internal Combustion Engine Vehicle and Battery Electric Vehicle Using LCA. Sustainability, 11(9), 2690.

Kim, B., Kang, J., Park, S., Jang, J., Hong, J.H., 2019. Study of unit cost estimation for the appropriate treatment of end-of-life photovoltaic (PV) module. New & Renewable Energy 2019. 3 Vol. 15, No. 1

Korea Environment Institute (KEI), 2018. Management status and improvement plans of waste solar panels

Latunussa, C.E.L., Ardente, F., Blengini, G.A., Mancini, L., 2016. Life Cycle Assessment of an innovative recycling process for crystalline silicon photovoltaic c. Sol. Energy Mater. Sol. Cells 156, 101–111.

Lunardi, M., Alvarez-Gaitan, J., Bilbao, J., Corkish, R., 2018. Comparative Life Cycle Assessment of End-of-Life Silicon Solar Photovoltaic Modules. Appl. Sci. 8, 1396.

Ministry of Environment (ME), 2018. The fifth final report for statistical survey of national wastes.

Paiano, A. (2015). Photovoltaic waste assessment in Italy. Renewable and Sustainable Energy Reviews, 41, 99–112.

Petrauskienė, K., Skvarnavičiūtė, M., & Dvarionienė, J. (2020). Comparative environmental life cycle assessment of electric and conventional vehicles in Lithuania. Journal of Cleaner Production, 246, 119042.

Winter, S., Rodríguez, C., Ciroth, A., Srocka, M., GreenDelta, 2014. openLCA Advanced Training. GreenDelta.

Xu QB, Mengjing Y, Kendall A, Wenzhi H, Guangming L, Schoenung JM (2013), Environmental and economic evaluation of cathode ray tube (CRT) funnel glass waste management options in the United States. Resour Conserv Recy 78:92–104

Chapter 6. Conclusions

6.1. Conclusions

The environmental impacts of EoL PV treatment were analyzed by LCA methodology of the ReCiPe endpoint and midpoint LCIA method. In this study, the EoL scenarios are firstly established, considering their applicability in Korea. Also, their environmental burdens and benefits caused by each treatment process and the material recovery, respectively, were compared for each scenario and treatment process. In the high-level EoL scenario, the treatment process of c-Si PV panels mainly composed of thermal and chemical treatment for delamination and material recovery, respectively, whereas the shredding for glass separation was a major process in the low-level EoL scenarios. All scenarios included the transport for collection, the disassembly of aluminum frame and junction box, and final disposal to landfill. It has been hypothesized that the material recovery from the EoL PV panels and their transportation would play a crucial role in determining the environmental dis- and burden during the recycling process. In the following, the most important results obtained from the analysis to validate this hypothesis are summarized.

The life-cycle environmental impacts of the EoL treatment for c-Si PV waste have been assessed by developing high- and low-level scenarios.

- The LCA results of EoL c-Si PV treatment without considering the avoided impacts by the materials recovery indicated that the impacts related to the high-level treatment were found to be higher than those of the low-level treatment because of higher environmental burdens resulted from the additional processes requiring chemicals, electricity, thermal treatment, etc. with respect to almost all impact categories.

- The impacts related to the toxicity indicators produced by the low-level treatment were higher than those by the high-level treatment because the hazardous components were not separated and shredded together with the cells for final disposal in the low-level treatment. In particular, the impact results of METP and FETP produced from the low-level treatment were almost 300 times higher than those from the high-level scenarios because of the emission of copper and silver ions to the groundwater.
- As a result of the endpoint analysis, the environmental benefits associated with the recovery of copper were most mainly contributed to the entire benefits in terms of the material recovery. When considering environmental disburdens related to the material recovery, the EoL treatment substantially avoided most of the environmental impacts in both high- and low-level recycling scenarios. The net environmental benefits associated with the high-level EoL treatment were found to be almost two times higher than those of low-level scenarios.
- The variability of the parameters related to the material recovery process influenced the environmental impacts in the range of -80% to +200%, while the variance of parameters associated with the transportation affected the environmental impacts from -40 to +58%. Therefore, for the precise assessment, the input parameters related to the transportation, i.e., distance, the type of vehicles, and transportation routes, should be based on more

realistic information.

- In general, the PV wastes are mainly treated at low level because of technical limitations and profitability. The recycling center, having a capacity of 3,600 ton/year for the high-level treatment, is now under construction in Korea. The amount of PV waste is expected to exceed this capacity after the year 2032. When constructing another recycling center, the economic benefits together with the environmental disburdens produced from the high-level treatment should be assessed.

The environmental impacts related to the metal recovery process were assessed for the three different leaching agents to choose the most environmentally friendly alternative.

- The environmental impacts of the chemical treatment resulted from the use of thiourea as a leaching agent were found to be superior in 9 out of 11 midpoint impact categories compared to those obtained by using HNO_3 and I_2-KI.
- Otherwise, the net benefit, the entire environmental impact caused by each treatment process and the material recovery, was higher when using HNO_3 as a leaching agent compared with two other leaching methods based on the level of endpoint analysis.

Environmental burdens were quantified with a focus on transportation during the treatment of PV wastes. The followings are the main results obtained in Chap.5.

- When the travel distance with the pre-treatment at the four collection centers was 362.3 km, which was longer than that of the scenario without the pre-treatment, 313.4 km, the results revealed that the impacts related to the transportation were reduced to the range of 73.8% to 85.8% in the high-level scenario. This mainly resulted from the reduction in the weight of the aluminum frame and the junction box and the less fuel consumption and emissions due to the utilization of a large-sized vehicle after being collected at the collection center.

- The amount of PV waste was predicted to be increased from 255 tons to 236,750 tons per year from the year 2025 to 2043, approximately by 929%, with an abrupt increase from 2037. Considering the PV waste projection, the GWP related to the transport of the PV waste without pre-treatment was found to be increased from 2.97×10^4 to 2.42×10^7 kg CO_2-eq in 2029 and 2043, respectively. This indicates that the transport of PV waste in 2043 contributes only 0.026% of the total national GHG emission from road transportation in 2017.
- According to the results of the comparative study on the transport scenarios, the reduction in the weight of the PV waste by the pre-treatment at the collection centers of future resource recycle has increased the environmental benefits. In addition, it is recommended to use a large-sized vehicle according to increase the amount of PV waste.
- Although the transport scenario with the pre-treatment at the collection centers showed the environmental benefits, the economic benefits would be considered in order to employ this scenario. The additional cost due to the increase in travel distance and the installation of equipment for the pre-

treatment in each collection center should be further evaluated in terms of the economic aspects.

Overall, the results obtained from this study strongly supported the hypothesis that material recovery and transportation could play a crucial role in determining the environmental impacts, especially in terms of the GWP, ODP, HTP, TETP, and MDP indicators.

6.2. Further studies

Because most of the input parameters used in this study were based on the literature and the assumptions, the calculated results are expected to be different from the environmental impacts obtained by the actual EoL treatment. The assumptions made in this study have been documented and validated with the LCA results, but further studies are necessary for a more precise assessment.

Further efforts are needed to assess the environmental impacts based on the field data, such as the amount of leaching agents, emissions, and recovered materials during the chemical treatment because the variabilities of these parameters are very high in the literature. In this study, the environmental impacts associated with the construction of the recycling facility have not been considered. Thus, the LCI related to its construction should be developed for more accurate and precise results. In addition, further studies should be conducted to analyze the combined environmental and economic benefits in order to expand the industrial market of high-level treatment.

국문 초록

전과정 평가를 통한 사용 후 태양광 패널 처리 방안에 대한 환경영향 비교 연구

2020년 6월 북극권 시베리아 지역의 기온이 38도를 기록하여 역대 최고 기온을 기록한 바 있다. 온실가스 중 이산화탄소는 지구온난화의 주범으로, 전세계적으로 에너지 부문이 58.8%를 기여하는 것으로 알려져 있다. 최근, 환경영향을 최소화하기 위해 재생에너지 보급을 확대하기 위한 노력이 이루어지고 있다. 태양광, 풍력, 바이오, 지열 등 재생에너지원은 화석연료를 대체하여 온실가스 배출을 저감하고 지구온난화를 완화할 대책으로 제시되고 있으며, 2050년에는 일차에너지의 20 - 40%가 신재생에너지를 통해 공급될 것으로 예측되고 있다.

신재생에너지 중 태양에너지는 가장 풍부하고 다양한 환경에서 이용이 가능한 에너지원으로, 태양전지를 이용하여 오염물질 배출이나 소음, 진동 없이 태양에너지를 전기에너지로 변환할 수 있다. 지난 10년간 태양광 시장은 급격히 증가하여, 2017년 전세계 누적 발전용량은 397.4 GW에 달하고 있다. 태양광패널의 수명이 25 - 30년임을 고려할 때, 태양광패널은 수 년 내로 환경 이슈가 될 것으로 예측되고 있다. 태양광패널의 누적 설치 용량과 태양광 설비 목표가 큰 나라는 특히 태양광 폐패널과 관련한 환경 부담에 직면할 것으로 예상된다. 2050년말 누적 폐패널 발생량은 6,000만톤-7,800만톤에 이를 것으로 전망된다.

전과정 평가 방법은 제품 또는 시스템의 제조, 사용, 폐기 등 모든 과정에 걸쳐 환경에 미치는 영향을 평가하는 기법이다. 몇 년간 생산과 운영 단계의 태양광 시스템에 대한 환경영향 평가가 이루어졌으나, 태양광 패널의 폐기 단계에 초점을 맞춘 연구는 제한적으로 이루어진 바 있다.

태양광 폐패널의 급격한 증가에 따라 폐패널의 적절한 처리 방안을 수립하는 것이 필요하다. 폐패널에 포함된 물질의 회수를 통해 환경영향을 줄일 수 있는 방안이 제안되고 있다. 폐패널의 처리과정은 폐패널의 처리시설로 수송하는 것으로 시작된다.

본 연구에서 폐패널을 열처리에 의한 물질분리와 화학처리에 의한 회수공정을 포함하여 높은 수준으로 처리할 경우 이에 수반되는 화학물질, 전기, 에너지 투입으로 인해 분리, 파쇄에 의한 물리적인 처리만을 포함하는 낮은 수준의 처리보다 환경부하가 더 큰 것으로 나타났다. 반면, 재활용을 통해 회수되는 물질에 의한 회피영향을 고려한 경우, 폐패널 처리에 의한 환경 부담은 두 경우 폐패널 처리에 의한 환경편익은 높은 수준으로 처리할 경우가 낮은 수준의 처리보다 두 배 정도 큰 것으로 나타났다.

본 연구를 통해 실질적으로 도출된 연구결과는 다음과 같다. 물질회수에 사용 가능한 세 가지 용출제 (질산, 요오드, 티오요소) 사용시의 환경영향을 비교한 결과, 티오요소를 사용한 경우, 화학처리 과정에서 발생하는 환경영향이 가장 작은 것으로 나타났다. 반면, 회수되는 물질을 고려하면 질산을 용출제로 사용한 경우 환경편익이 다른 두 가지 용출제의 경우보다 큰 것으로 나타났다. 태양광 폐패널 수송시 수거센터에서 전처리를 거치는 경우 수송에 의한 환경부하는 전체 수송거리의 증가에도 불구하고 재활용 시설로 직접 수송하는 경우보다 14.2%에서 26.2%로 감소하는 것으로 나타났다. 이는 전처리에 의한 폐패널 무게의 감소 및 대형차량 사용으로 인한 수송 효율성의 증가 때문으로 분석되었다.

회수된 재활용 물질에 의한 생산 회피를 고려하더라도 환경영향 범주 중 일부는 처리과정에 의해 환경부하가 발생한다. 결론적으로, 폐패널의 처리시의 경제적, 환경부하의 감소를 위해서는 자원회수와 수송에 대한 고려가 필요하다.

주요어 : 폐기물 처리, LCA, PV패널, 자원회수, 수송

학번 : 2011-30270