Construction Packaging Waste Management

A case study of LDPE packaging waste in Victoria

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Executive Summary

Low-Density Polyethylene (LDPE) is widely used in the construction industry for packaging, protecting materials, and facilitating their handling and transportation. However, its use typically follows a linear 'take-make-dispose' model that results in significant environmental impacts. Properly managing this waste stream is essential, as it can reduce the environmental impacts of construction activities. One key step is ensuring LDPE remains within the economy through effective End of Life (EoL) management to support a Circular Economy (CE).

This study employed Life Cycle Assessment (LCA) to assess the Greenhouse Gas (GHG) emissions associated with three alternative EoL management scenarios for LDPE used as packaging for construction materials—waste-to-energy, mechanical recycling and chemical recycling—compared to the business-as-usual practice of disposal in landfill.

Mechanical recycling, with total GHG emissions of 6.10 kgCO_2-e/kg, emerged as the most favourable option, followed by chemical recycling at 7.46 kgCO_2-e/kg. Considering the offset that can be achieved by the avoided virgin polymer production in these scenarios, they present significant advantages compared to disposal in landfill. The suitability of these two pathways, however, depends on waste characteristics, with factors such as contamination, mixing with other waste, and the need for washing and sorting affecting both the choice of pathway and overall emissions. The waste-to-energy scenario resulted in the highest GHG emissions, particularly as the ongoing decarbonisation of national electricity grids diminishes the credit that would otherwise have been attributed to avoided electricity generation. Additionally, among all activities within the life cycle, the production of LDPE packaging from virgin polymer accounted for over 50% of the total GHG emissions across all scenarios, highlighting the significance of this stage.

The findings provide valuable recommendations for practitioners and policymakers in developing best practices for the life cycle management of LDPE packaging in construction, ultimately contributing to a CE and reduced GHG emissions.

Structure

This report is structured in five sections as follows:

1. **Introduction** sets the stage by introducing the problem of construction LDPE packaging waste, highlighting its environmental impacts and the challenges associated with its management within the construction industry. The section then outlines the aim and objectives of this study.

2. **Background** provides an overview of plastics in Australia, reviews policy instruments aimed at creating a CE for plastics and explores management options for LDPE waste.

3. **Research Design** explains the methodological framework of the study, detailing the LCA research design employed to evaluate the environmental impacts of LDPE waste management scenarios. The section describes the functional unit, system boundary and quantification method for each activity within the life cycle.

4. **Findings and discussion** present the results of the LCA for four scenarios and analyse their respective trade-offs. The section also interprets the findings and examines their broader implications for waste management strategies in the construction sector.

5. **Concluding remarks** summarise the findings and provide recommendations on how these insights can inform decision-making in practice.

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List of Acronyms

AusLCI	Australian National Life Cycle Inventory Database
CE	Circular Economy
EoL	End of Life
GHG	Greenhouse Gas
HTL	Hydrothermal Liquefaction
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LDPE	Low-Density Polyethylene
PW	Plastic Waste
EPA	Environment Protection Authority

1. Introduction

Low-Density Polyethylene (LDPE) is widely used in the construction industry for packing materials such as clay bricks or steel coils to preserve the quality of construction materials and simplify their transport (Pešta et al., 2020). However, LDPE packaging is typically short-lived compared to construction materials, resulting in a significant amount of waste during construction activities. **In Australia, the construction industry alone generated 54,996 tonnes of LDPE waste in 2018–2019, the majority of which ended up in landfill** (Hossain et al., 2022). Disposing of LDPE in landfill not only wastes the resources embedded in its production but also presents serious risks to both environmental and human health given its prolonged degradation period. Therefore, proper management of this waste material is critically important.

The challenge of LDPE waste management reflects broader issues in the global plastics economy. While plastics offer undeniable benefits, they remain part of a predominantly linear economy that follows a 'take-make-dispose' model. This model is characterised by growing consumption and limited recovery, leading to the generation of massive amounts of waste. Plastic use has surged from 1.5 Mt in 1950 to 460 Mt in 2019, with projections suggesting this figure will nearly triple by 2060 (OECD, 2022). The rise in plastic use is expected to be mirrored by a corresponding tripling of Plastic Waste (PW), with nearly half of it still being disposed of in landfill if business continues as usual (OECD, 2022). **Plastics now account for 12% of total global waste by weight, the vast majority of which ends up in landfill** (Hossain et al., 2022). A substantial portion of this PW is derived from polyolefins, including LDPE (Yang et al., 2022).

In recent years, the concept of a Circular Economy (CE) has gained traction as a potential solution. A CE fosters the efficient use of resources by creating cyclical supply chains, in which the notion of waste is eliminated. By treating the End of Life (EoL) of products as a resource, a CE links waste management to resource circulation, ensuring that valuable materials remain in the economy while supporting environmental sustainability. **For plastics such as LDPE, a CE involves reuse or recycling at their EoL to move away from the traditional linear 'take-make-dispose' model**. However, the CE for plastics is still in its infancy, partly due to the low cost of polymers and their varying additives.

The Ellen MacArthur Foundation, in its "The New Plastics Economy" report, highlights the importance of creating an effective after-use plastics economy to promote increased recycling rates of (packaging) polymers (EMF, 2017). By doing so, it is argued that recycling rates can increase, resource productivity can improve and the leakage of PW into the environment can be minimised.

However, to improve the environmental performance of LDPE packaging waste management, it is essential to evaluate the environmental impacts of EoL management options**. Life Cycle Assessment (LCA) is a standard method for investigating the environmental impacts of a product throughout its life cycle which has been increasingly used over the past decades** (Rebizer et al., 2004). By employing LCA, one can assess the environmental impacts of LDPE throughout its life cycle and gain insights into the factors that influence the choice of EoL management options. The findings of such an analysis could support decision-making processes aimed at reducing environmental impacts.

While previous LCA studies have examined alternative packaging materials, they often featured varying functional units and system boundaries, primarily focusing on packaging used in the food processing industry (Gómez & Escobar, 2022). This study aims to provide a comprehensive analysis of the environmental impacts associated with EoL management of LDPE used as packaging for construction materials. **The specific objectives of this study are as follows:**

1. To quantify the life cycle GHG emissions associated with LDPE used as construction packaging across four EoL management scenarios: disposal in landfill (business-as-usual), waste-to-energy, mechanical recycling and chemical recycling
2. To identify the alternative that results in the lowest GHG emissions

2. Background

This section provides a background on plastics in Australia, policy instruments aimed at creating a CE for plastics as well as management options for LDPE packaging waste.

2.1 Plastics in Australia

Plastics are synthetic materials made from polymers, which are long chains of molecules derived primarily from fossil fuels like crude oil and natural gas. Plastic production begins with the extraction of these raw materials, which are then refined into smaller building blocks such as ethylene and propylene. These molecules are chemically bonded into polymers through a process called polymerisation, resulting in a versatile material that can be moulded into a wide range of products (Laredo et al., 2023). **Despite their utility, the widespread consumption and resistance to degradation of plastics have raised significant environmental concerns**, as much of the plastic produced ends up in landfills or pollutes natural ecosystems.

In Australia, plastic consumption has surged by 166% since 2000, rising from 1.5 Mt to 3.5 Mt by 2018–19 (O'Farrell, 2020). Despite this increase, PW remains the least recovered waste stream, with a recovery rate of just 11.5% (O'Farrell, 2020). **Figure 1** shows Australian plastics consumption and recovery by polymer type in 2018–19. Although LDPE was not the most consumed polymer that year, it still represented a substantial 351,900 tonnes.

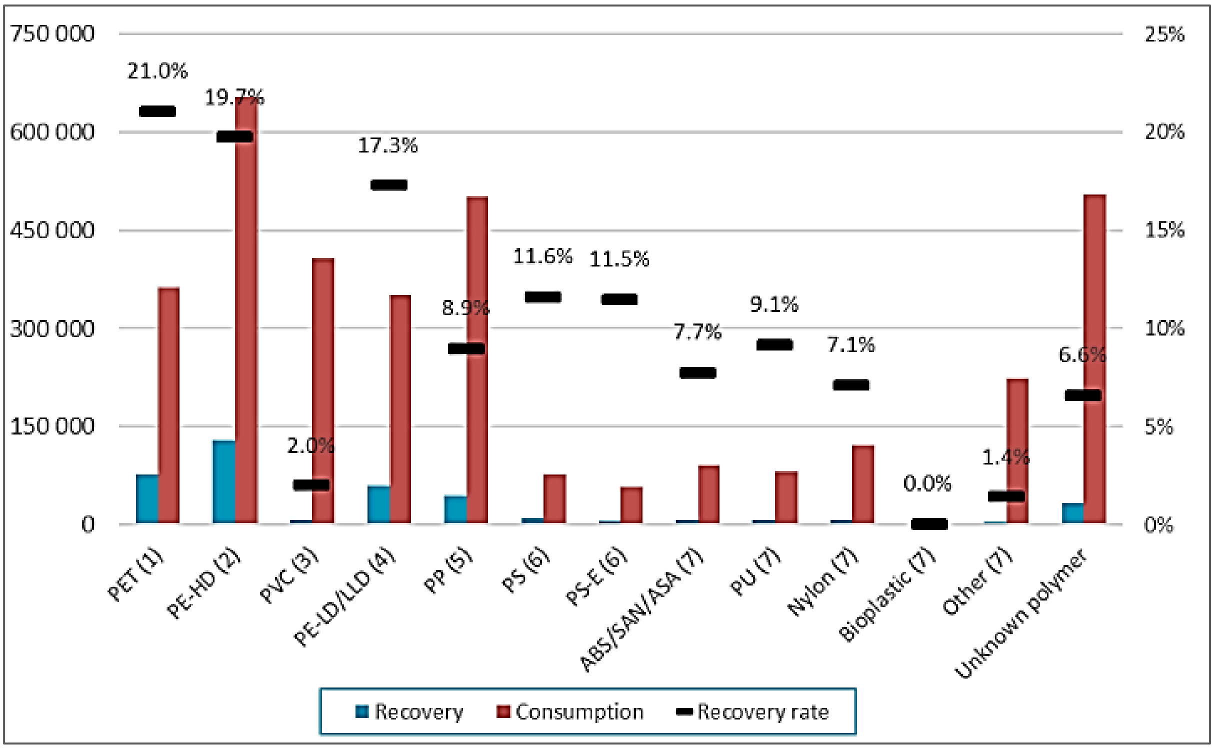

Figure 1. Australian plastics consumption and recovery by polymer type in 2018–19

Source: O'Farrell (2020)

Australia's polymer manufacturing industry primarily depends on imported resins, though LDPE remains one of the few resin types still produced domestically. **LDPE is extensively used in packaging and ranks among the leading contributors to PW in Australia**. In 2018–19, LDPE waste generation reached 0.42 Mt, with major contributions from the construction industry (54,996 tonnes), manufacturing (121,746 tonnes), and other sectors (187,593 tonnes) (Hossain et al., 2022; O'Farrell, 2020). Victoria achieved the highest overall recovery rate at 16%, with LDPE specifically recovering at 27.8% (O'Farrell, 2020).

2.2 Policy frameworks in circularity of plastics

National policies, laws and international agreements have become pivotal in fostering a CE for plastics. In Australia, the National Waste Policy Action Plan (NWPAP, 2019) lays out a strategic framework to address challenges within the waste and resource recovery sectors. Key targets include banning PW export, achieving an 80% average resource recovery rate, increasing the use of recycled content, and phasing out problematic and unnecessary plastics. Complementing this, Australia’s Circular Economy Framework (DCCEEW, 2024) sets an ambitious goal of doubling the country’s circularity by 2035. This includes reducing per capita material footprint by 10%, increasing material productivity by 30%, and safely recovering 80% of resources. It also highlights circular packaging as a priority area.

As shown in **Figure 2**, **Australia has also set ambitious targets under the National Packaging Covenant Strategic Plan (APCO, 2019), which aims for 100% of packaging to be reusable, recyclable, or compostable**. Additionally, 70% of plastic packaging is to be recycled or composted, with 50% of the average recycled content to be included in packaging—an increase from 30% in 2020. The plan also focuses on eliminating problematic and unnecessary single-use plastic packaging. These targets are backed by industry, the federal government and all Australian state and territory governments, as outlined in both the 2019 National Waste Policy Action Plan (NWPAP, 2019) and the 2021 National Plastics Plan (DAWE, 2021).

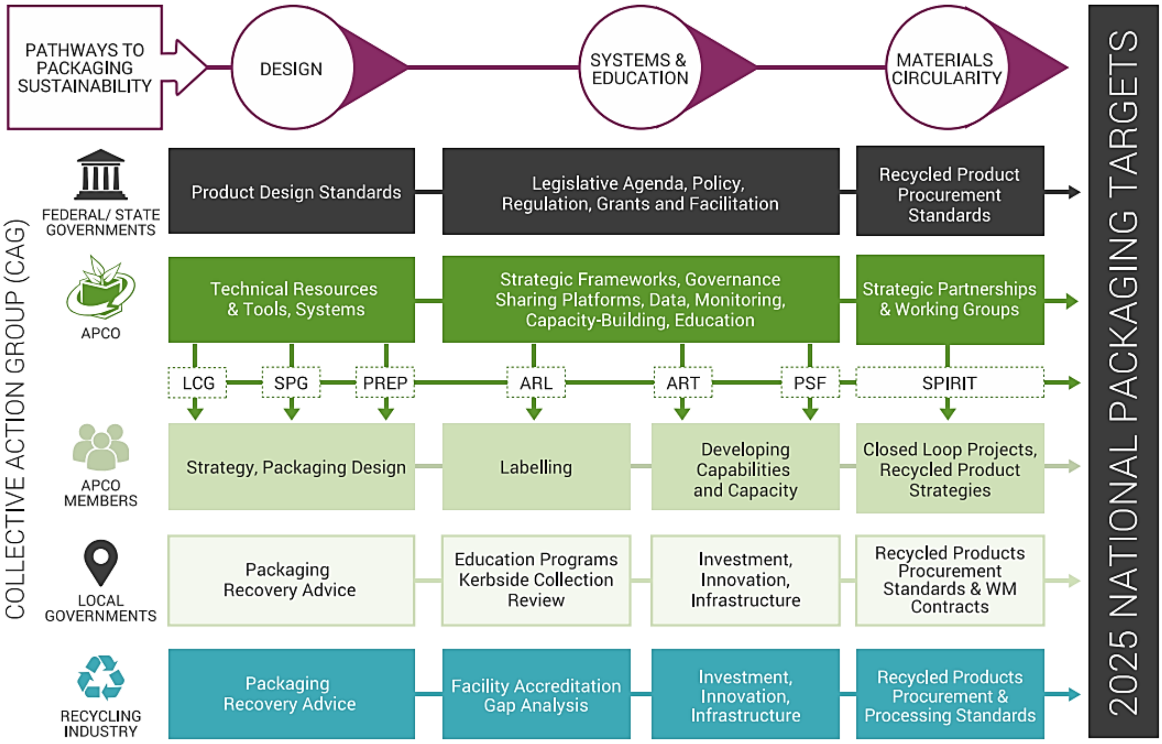

Figure 2. Sustainable packaging pathway and stakeholder activities
Source: APCO (2019)

In Victoria, the Waste to Energy Framework recognises waste-to-energy as a final opportunity to extract value from materials that would otherwise be destined for landfill, following efforts to avoid, reuse or recycle waste (DELWP, 2021). It establishes regulations to manage waste-to-energy practices, including placing a cap on the total amount of waste that can be heat-treated to generate energy.

2.3 Overview of LDPE waste management options

As shown in **Figure 3**, multiple EoL management options exist for plastics, including LDPE. These include recycling (mechanical, chemical or biological), incineration (with or without energy recovery) and landfilling (Hossain et al., 2022). However, the use and scale of these options vary considerably across nations, depending on their available infrastructure and regulatory frameworks.

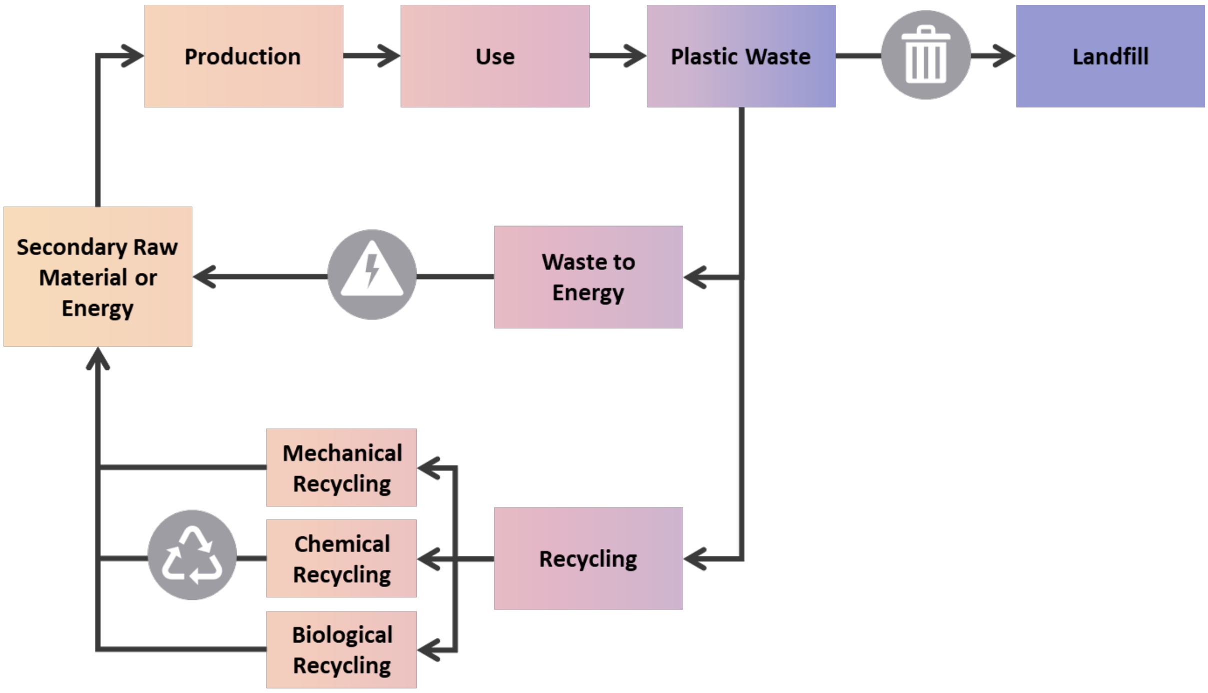

Figure 3. End-of-life pathways of plastics
Source: Authors

In Australia, PW has predominantly been landfilled. According to the Australian Plastics Recycling Survey, only about 11.5% of PW is recovered, leaving the overwhelming majority disposed of in landfill (O'Farrell, 2020). In Victoria, during the 2017–18 biennium, PW amounted to 585,200 tonnes. Of this, a significant 448,000 tonnes (76.6%) were disposed of in landfills, while 130,000 tonnes (22.2%) were recycled through mechanical processes, and 7,200 tonnes were incinerated for energy recovery (Santos et al., 2021).

However, **Australia's approach to PW management is undergoing significant changes, driven by increasing investments in recycling and energy recovery infrastructure**. By 2025, the country is set to benefit from new mechanical recycling facilities with a combined capacity of 300,000 tonnes per year, alongside chemical recycling plants capable of handling an additional 200,000 to 300,000 tonnes annually. Waste-to-energy infrastructure is also advancing, with plastics serving as a key contributor to the energy value of waste streams (O'Farrell & Pickin, 2023). Among all states, Victoria leads with the highest number of facilities dedicated to processing various types of plastic (Hossain et al., 2022). However, no facilities in Australia currently process bioplastics (Hossain et al., 2022).

With Victoria's evolving waste management infrastructure, three EoL management options present themselves as viable alternatives to the business-as-usual approach of disposing of construction LDPE packaging waste in landfills: **waste-to-energy**, **mechanical recycling** and **chemical recycling**. The details of each are discussed further below.

2.3.1 Disposal in landfill

Disposal in landfills is the dominant method of managing PW, including LDPE, within Australia's waste management system. This business-as-usual approach is reinforced by the economic incentives underpinning PW management in Australia. **Skip bin companies that collect waste typically charge by volume ($/m³), while disposal costs, including Environment Protection Authority (EPA) landfill levies, are based on weight ($/t). This pricing structure makes it more cost-effective to dispose of lighter waste in landfills**. Consequently, after recovering heavier materials like concrete and steel, which have resale value, lighter materials like LDPE waste are disposed of due to the absence of financial incentives for recovery.

Landfills are typically designed to operate for over 20 years, during which time PW undergoes five stages of stabilisation influenced by geological, hydrological, biological and thermal processes. However, research has revealed that landfills are a significant source of microplastics, which gradually leak into the surrounding environment and cause considerable harm to both environmental and human health (Hossain et al., 2022).

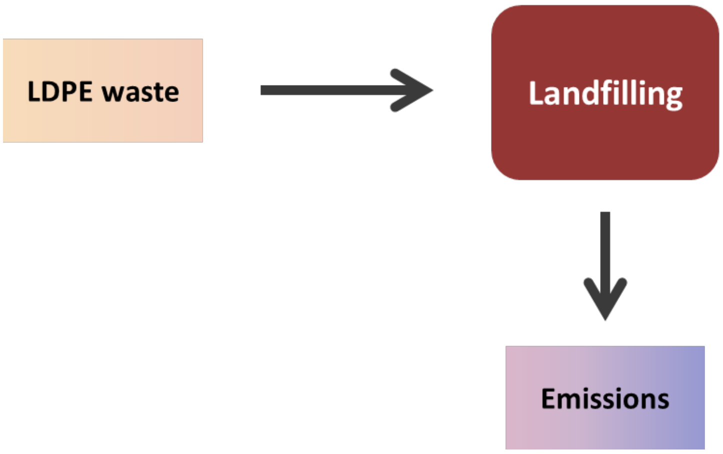

Figure 4. Disposal in landfill system boundary
Source: Authors

2.3.2 Waste-to-energy

Waste-to-energy is a process that transforms waste materials into usable forms of energy, such as electricity, heat, or fuel, through methods like combustion, gasification, or anaerobic digestion. Plastics, in particular, offer a high energy yield comparable to those of fossil fuels, 43.3 MJ/kg for polyethylene as opposed to 45.2 MJ/kg for gas oil (Baytekin et al., 2013). This makes plastics a cost-effective energy source. Although waste-to-energy does not eliminate the need for landfills entirely, it greatly reduces the volume of waste destined for landfills.

In Victoria, three incineration plants are planned, with a combined capacity to process nearly 1.5 Mt of kerbside waste annually, where the plastic fraction is expected to contribute significantly to the energy value of waste streams. Additionally, two other facilities using gasification technology are being developed, together capable of managing roughly one-third of the 5 Mt of waste currently sent to Victorian landfills each year (Tippet & Schapova, 2024).

The incineration process involves burning waste in large furnaces. The heat generated powers boilers to produce steam. The steam then drives turbines to produce energy. This energy can be used to power the incineration facility itself and supply electricity to the state grid. The process generates several by-products, including bottom ash, a non-combustible residue that can be sorted to recover recyclable materials or sent to landfill, and fly ash, along with other emissions, which pass through the facility's smokestacks, where they are captured at various stages and either recycled or disposed of. The incineration of plastics requires advanced pollution control measures, as the process yields toxic and noxious dioxins that must be carefully monitored (**Figure 5**).

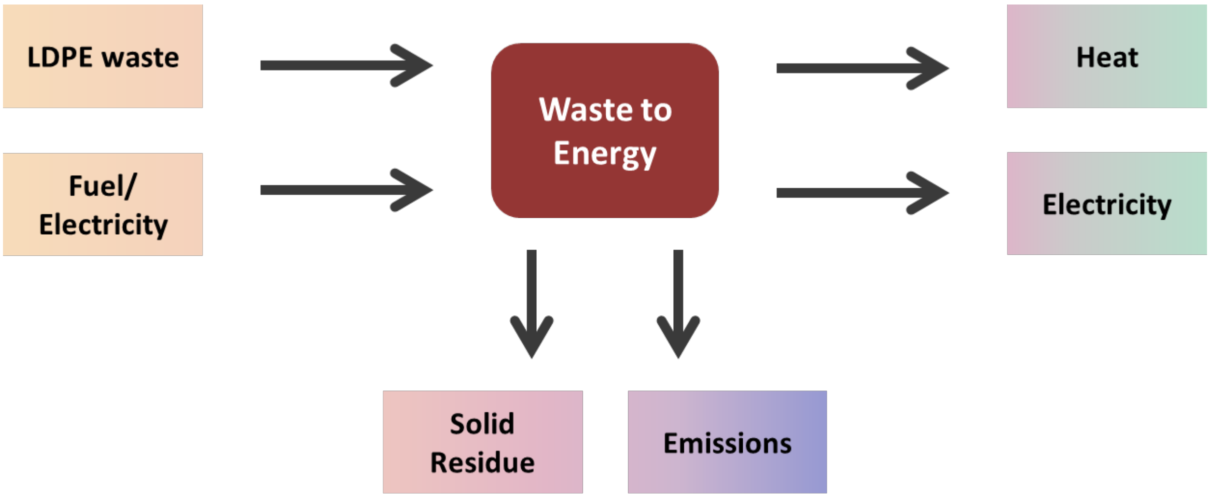

Figure 5. Waste-to-energy system boundary
Source: Authors

2.3.3 Mechanical recycling

Mechanical recycling is the most widely used method for managing PW. It involves converting PW into 'new' (secondary) raw materials without altering its basic structure (PlasticsEurope, 2019).

In Victoria, about 12 facilities are equipped to process LDPE (Hossain et al., 2022). The process typically consists of several steps, which may occur in varying sequences, be repeated multiple times, or even be skipped altogether, depending on the composition and origin of the waste. These steps may involve: separating and sorting materials based on parameters such as shape, density, size, colour or chemical composition; baling the plastics to ease transportation when processing does not take place at the sorting facility; dry-cleaning or washing to eliminate contaminants; grinding to break down plastic items into smaller flakes; and pelletising, where flakes may be reprocessed into granules through methods such as extrusion or injection moulding, making them more suitable for further use by manufacturers (Ragaert et al., 2017). The output of mechanical recycling includes products like flakes and pellets, which can serve as secondary raw materials (**Figure 6**).

However, **for plastics to undergo mechanical recycling, they must first be sorted and rid of contaminants, such as coatings and paints**. Various separation techniques have been developed such as dry or wet gravity separation, electronic or magnetic density separation, flotation, and sensor-based sorting together with auxiliary segregation techniques such as magnetic or eddy-current separation (Beghetto et al., 2021). Sorting is particularly difficult for soft plastics and unfeasible for multilayer plastic packaging.

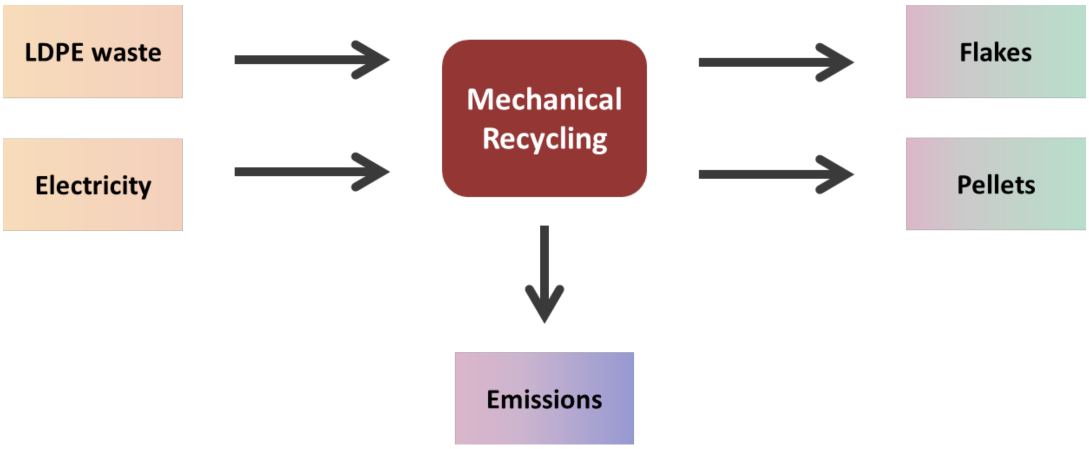

Figure 6. Mechanical recycling system boundary
Source: Authors

2.3.4 Chemical recycling

Chemical recycling involves breaking down the plastics back into their monomer structure, enabling the production of new materials based on the molecules that have the potential for a wide range of applications. **This process reduces the need for virgin inputs in the production of fuels and plastics**.

This recycling method is particularly advantageous for handling heterogeneous and contaminated PW where separation is neither economical nor technically feasible (Ragaert et al., 2017). For example, soft plastics, such as LDPE packaging, are challenging to recycle due to the presence of fillers, colouring agents and multiple polymer types. These characteristics limit the effectiveness of mechanical recycling and often result in such plastics being landfilled.

Chemical recycling can be divided into chemolysis and thermochemical recycling. Chemolysis includes alcoholysis, hydrolysis, glycolysis, aminolysis, solvolysis, catalysis, organic catalysis, enzymatic and hydrolysis (Beghetto et al., 2021). Thermochemical recycling encompasses pyrolysis which breaks down PW by heating it in the absence of oxygen; gasification which converts PW into syngas (a mixture of hydrogen, carbon monoxide and methane) at high temperatures using limited oxygen or steam; and Hydrothermal Liquefaction (HTL) which uses water under high temperatures and pressure to transform PW into oil similar to refined fossil crude oil (Yang et al., 2022).

Among these recycling technologies, HTL is a modern alternative for plastic recycling. In Victoria, plans are underway to establish an advanced recycling facility utilising HTL technology, which will initially process approximately 20,000 tonnes of EoL plastic per year, with a projected capacity to scale up to 120,000 tonnes annually (Licella, 2023).

The HTL process begins with the preparation of PW, where non-plastic contaminants are removed, leaving mixed EoL plastics ready for processing. The prepared plastic is then melted and pressurised, after which it is mixed with water at high temperature and pressure. This mixture is then fed into the reactors, where the plastic's chemical structure is broken down. Following the reaction, the mixture is depressurised to stabilise the outputs. The process separates the resulting materials into Plasticrude and gaseous by-products, the latter being used to generate energy for the facility or flared if not needed. Finally, the Plasticrude is stored as a valuable resource for further processing into fuels or new plastics (**Figure 7**).

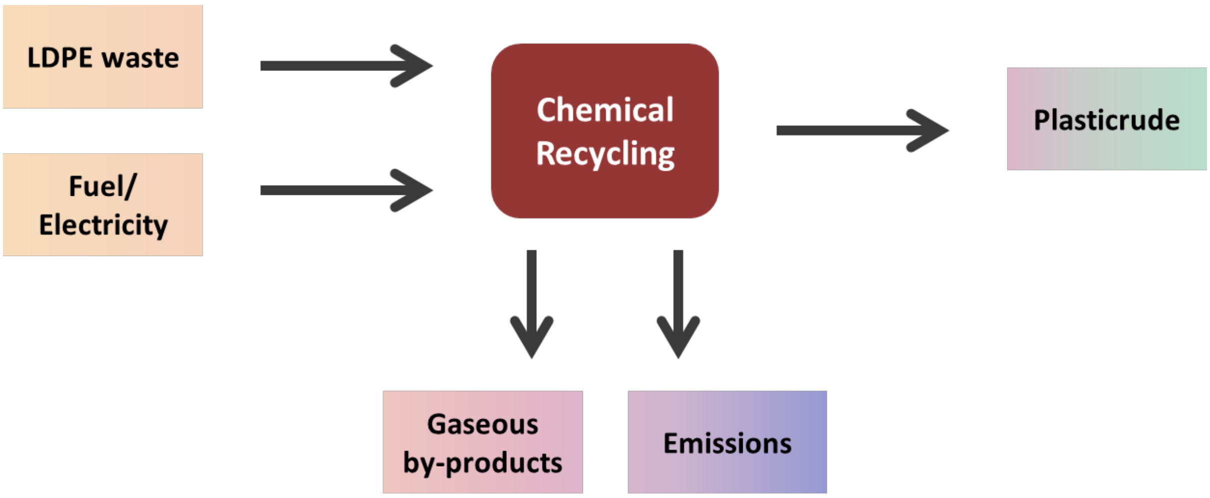

Figure 7. Chemical recycling system boundary
Source: Authors

3. Research Approach

This study applied LCA to assess the GHG emissions associated with four EoL management scenarios of LDPE used as packaging for construction materials. LCA is a standard method for quantifying the environmental impacts of a given product across different stages of its life cycle (Rebizer et al., 2004). As shown in Figure 8, this study adopts ISO 14040 (2006) which establishes a four-step framework for conducting an LCA:

1. **Goal and Scope Definition:** This initial step outlines the goal of the assessment and its boundaries.
2. **Life Cycle Inventory (LCI) Analysis:** This step involves collecting data and performing calculations to quantify the inputs and outputs associated with the system under study.
3. **Life Cycle Impact Assessment:** In this step, the LCI results are linked to environmental impact indicators and categories, allowing for an evaluation of their significance.
4. **Interpretation:** The final step entails checking for completeness and consistency, as well as assessing the sensitivity, accuracy and uncertainty of the results obtained.

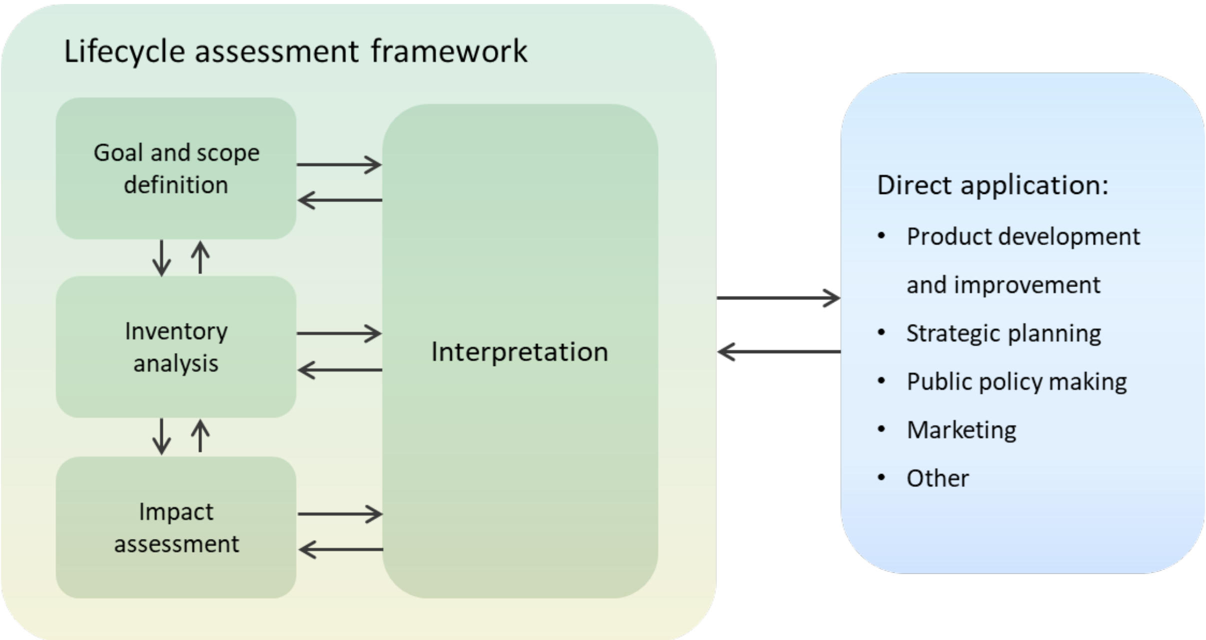

Figure 8. Steps and applications of an LCA
Source: ISO 14040 (2006)

3.1 Goal and scope definition

The goal of this study is to quantify the life cycle GHG emissions associated with LDPE used as construction packaging across four different EoL management options, in order to identify the alternative that results in the lowest GHG emissions.

3.1.1 Functional unit

The functional unit of this study is defined as 1 kilogram of LDPE. This unit serves as a reference point, allowing for a consistent comparison of results across different EoL management options. The GHG emissions associated with this functional unit are measured in terms of kilograms of carbon dioxide per kilogram of LDPE ($kgCO_2$-e/kg).

3.1.2 System boundary

As shown in **Figure 9**, the system boundary of this study includes LDPE packaging production, the transportation of the packaging to the construction material manufacturer, the transportation of packed materials to the construction site, the transportation of packaging waste from the construction site and the EoL management process of each scenario. It excludes the GHG emissions associated with the production of construction material, its packing and the construction process itself as these emissions are not pertinent to the life cycle of the packaging material.

The geographical boundary of this study is Victoria, Australia.

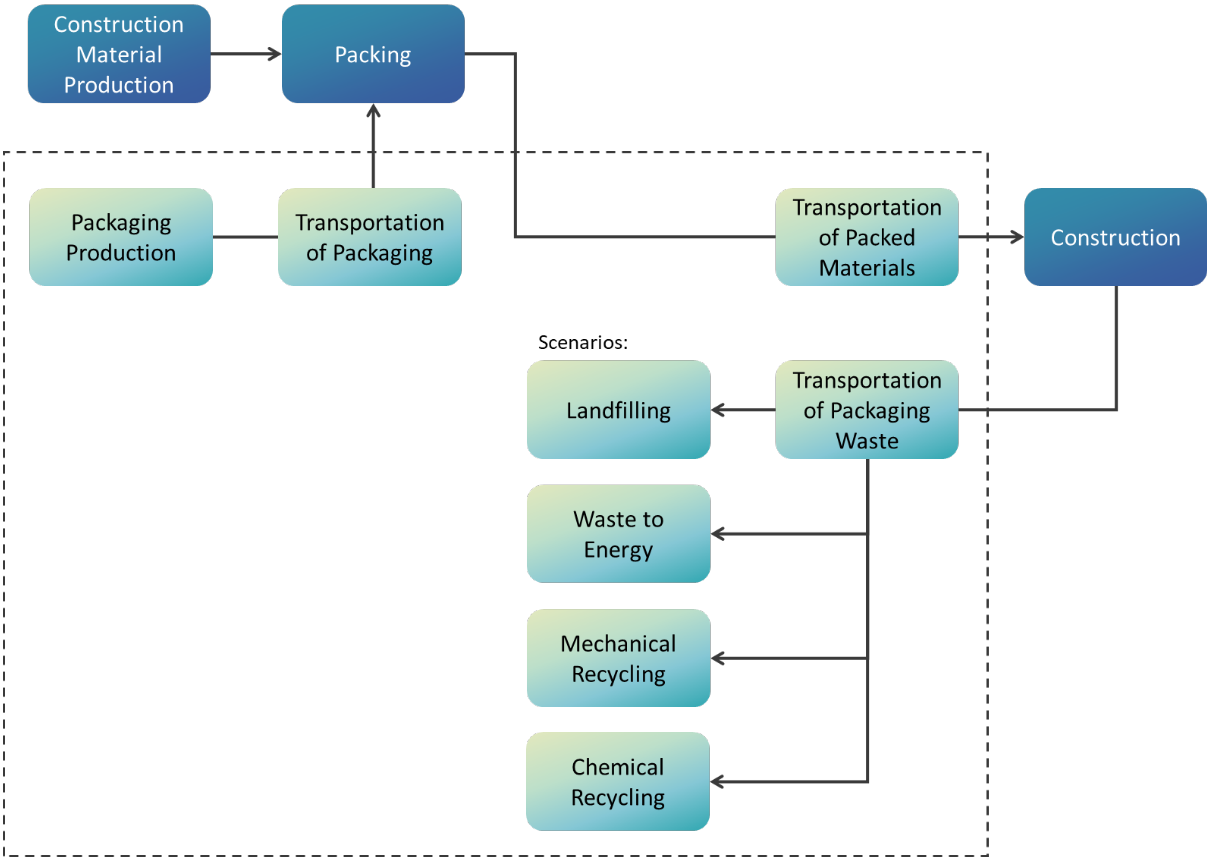

Figure 9. The system boundary of the study
Source: Authors

3.2 Quantifying greenhouse gas emissions

The LCI for estimating GHG emissions across the scenarios indicated above was developed using data from various sources including Australian government reports, established databases and existing literature as outlined below.

3.2.1 Packaging production

The LDPE packaging production GHG emissions include emissions associated with the extraction, transportation and manufacturing of raw materials. It is assumed that no recycled content is used in the production of the LDPE packaging. Hence, as shown in **Table 1** the production GHG emissions of the LDPE packaging were quantified using the Australian National Life Cycle Inventory Database (AusLCI) database (Grant, 2023). The GHG emissions coefficient represents the total GHG emissions associated with the extraction of raw material through the factory gate per unit of material.

Table 1. GHG emissions coefficient of LDPE.

Item	Value	Unit
LDPE granulate GHG emissions coefficient	3.60[1]	kgCO2-e/kg
LDPE film GHG emissions coefficient	4.31[1]	kgCO2-e/kg

1. based on AusLCI database V1.42 (Grant, 2023).

3.2.2 Transportation of packaging

The GHG emissions from the transportation of LDPE packaging encompass emissions generated during the transport of LDPE packaging from the manufacturing site to the construction material manufacturing facility. It is assumed that diesel-fuelled articulated trucks are used to distribute packaging materials to construction material manufacturers. As summarised in Table 2, the transport GHG emissions factor was extracted from AusLCI database V1.42 (Grant, 2023). Travel distances were calculated using Google Maps, based on the distance between preferred manufacturers, all located within Victoria. Outward and return trips were assumed to cover the same distance.

Table 2. Transportation of packaging process data.

Item	Value	Unit
Transport GHG emissions factor	0.276[1]	kgCO2-e/tkm
Travel distance	80.00	km

1. based on AusLCI database V1.42 (Grant, 2023)

3.2.3 Transportation of packed materials

The GHG emissions from the transportation of packed materials include those generated during the transport of materials from the manufacturing site to the construction site. These emissions, however, vary depending on the specific construction material for which LDPE is used as packaging, the type of vehicle used for transportation and the proportion of transportation emissions attributable to LDPE packaging. For this study, it is assumed that 10% of the load consists of packaging material and that the construction site is located in Melton, Victoria. Diesel-fuelled trucks with a capacity of up to 30 tonnes are used for transporting packed materials. Travel distances were calculated using Google Maps, assuming that outward and return trips cover the same distance (**Table 3**).

Table 3. Transportation of packed materials process data.

Item	Value	Unit
Transport GHG emissions factor	0.308[1]	kgCO2-e/tkm
Travel distance	115.00	km

1. based on AusLCI database V1.42 (Grant, 2023).

3.2.4 Transportation of packaging waste

The GHG emissions associated with the transportation of LDPE packaging waste include those generated during the transport of waste from construction sites to landfills or processing facilities. The waste collection process involves a diesel-fuelled truck leaving the depot with empty bins or skips and travelling to construction sites. The truck collects full bins or skips from multiple sites and, if necessary, swaps them with empty ones. The collected waste is then delivered either to a processing facility or disposed of in a landfill. Following delivery, the truck returns to the depot. In this process, two types of trucks were considered: larger trucks for transporting waste to landfills and smaller trucks for delivery to processing partners.

Similar to Section 3.2.2, the transport GHG emissions factor was extracted from the AusLCI database V1.42 (Grant, 2023). The travel distances were measured using Google Maps based on the location of the preferred landfill or processing facility, depending on the specific waste management scenario, with all facilities selected within Victoria. When calculating travel distances, the proposed locations for new energy recovery and chemical recycling facilities were considered (**Table 4**).

Table 4. Transportation of packaging waste process data.

Item	Value	Unit
Transport GHG emissions factor (16 to 28t truck)	0.308[1]	kgCO2-e/tkm
Transport GHG emissions factor (3, 5 to 16t truck)	0.276[1]	kgCO2-e/tkm
Travel distance to landfill	162.00	km
Travel distance to an energy recovery facility	54.00	km
Travel distance to a mechanical recycling facility	34.00	km
Travel distance to a chemical recycling facility	34.00	km

1. based on AusLCI database V1.42 (Grant, 2023).

3.2.5 Disposal in landfill

The GHG emissions associated with disposing of LDPE packaging waste in landfills include those generated during the processing of waste for landfilling and its subsequent decomposition. However, LDPE is classified as inert waste, containing minimal degradable organic carbon. Hence, while decomposition may occur over extended periods, the process is generally considered negligible. Consequently, methane conversions and emissions factors are assumed to be zero for this study (**Table 5**).

Table 5. Disposal in landfill process data.

Item	Value	Unit
Waste treatment emission factor	0.01[1]	kgCO2-e/kg
Waste conversion factor	0.00	kgCO2-e/kg

1. based on AusLCI database V1.42 (Grant, 2023).

3.2.6 Waste-to-energy

Waste-to-energy GHG emissions include emissions from the preparation of LDPE packaging waste for incineration (e.g., sorting) and the incineration process itself. It is assumed that no waste processing occurs prior to incineration. As summarised in Table 6, the heating value and plant electricity use were sourced from O'Farrell and Pickin (2023). Victoria's electricity GHG emissions factor was obtained from NGA (2022). Additionally, it is assumed that no external fuel input is required for normal operations, and no heat export credit is applied for the treatment of LDPE packaging waste. During combustion, the polymers are expected to undergo near-complete oxidation, resulting in negligible solid residues. A credit is applied for the electricity generated during the incineration process, as it offsets the need for electricity from alternative energy sources. Credits for avoided heat were excluded due to insufficient market demand in Victoria (Demetrious & Crossin, 2019).

Table 6. Waste-to-energy process data.

Item	Value	Unit
Heating value	44.60[1]	GJ/t
Plant electricity use	260.00[1]	kWh/t
Electricity GHG emissions factor (Victoria)	0.92[2]	kgCO2-e/kWh
Electricity export	2200.00[1]	kWh/t

1. based on O'Farrell and Pickin (2023)
2. emissions from consumption of purchased electricity from a grid, based on NGA (2022)

3.2.7 Mechanical recycling

Mechanical recycling GHG emissions include emissions generated during the sorting of LDPE packaging waste and its subsequent mechanical recycling processes. As shown in Table 7, the electricity usage values for sorting and mechanical recycling into virgin-equivalent flakes and pellets are derived from O'Farrell and Pickin (2023). The electricity GHG emissions factor for Victoria was sourced from NGA (2022). It is assumed that both sorting and mechanical recycling are conducted within the same facility. Additionally, a 50/50 split was considered for the proportion of mechanically recycled LDPE waste converted into flakes and pellets.

Table 7. Mechanical recycling process data.

Item	Value	Unit
Sorting electricity use	17.00[1]	kWh/t
Mechanical recycling into virgin equivalent flakes electricity use	1,480.00[1]	kWh/t
Mechanical recycling into virgin equivalent pellets electricity use	2,230.00[1]	kWh/t
Electricity GHG emissions factor (Victoria)	0.92[2]	kgCO2-e/kWh

1. based on O'Farrell and Pickin (2023).
2. emissions from consumption of purchased electricity from a grid, based on NGA (2022).

3.2.8 Chemical recycling

Chemical recycling GHG emissions include emissions generated during the sorting of LDPE packaging waste and its subsequent chemical recycling processes. As outlined in Table 8, the electricity usage for sorting, front-end pre-processing, reactor operations, and the conversion of liquid plasticrude to ethene is based on O'Farrell and Pickin (2023). It is assumed that sorting and chemical recycling occur within the same facility. Of the incoming plastic, 85% is expected to be recovered into plastic crude, which is then sent for downstream processing into fuels or plastics. The remaining 15% is recovered as a gaseous co-product, which is either combusted for facility energy generation or flared.

Table 8. Chemical recycling process data.

Item	Value	Unit
Sorting electricity use	17.00[1]	kWh/t
Front-end pre-processing electricity use	80.00[1]	kWh/t
Reactor electricity use	210.00[1]	kWh/t
Plasticrude to ethene conversion electricity use	412.00[1]	kWh/t
Plasticrude to ethene conversion fuel use	35.60[1]	GJ/t
Fuel use GHGE factor	54.80[2]	kg CO2-e/GJ
Electricity GHGE factor (Victoria)	0.92[2]	kgCO2-e/kWh

1. based on (O'Farrell & Pickin, 2023).
2. emissions from consumption of purchased electricity from a grid, based on (NGA, 2022).

4. Findings and Discussion

This study applied LCA to evaluate the GHG emissions associated with four EoL management scenarios for construction LDPE packaging: disposal in landfill (business-as-usual), waste-to-energy, mechanical recycling and chemical recycling. By conducting a detailed process analysis, the study quantified emissions across the life cycle. Figure 10 presents the results of the analysis. Emissions are disaggregated by activity to enable a comparison of the contributions from production, transportation and EoL management processes.

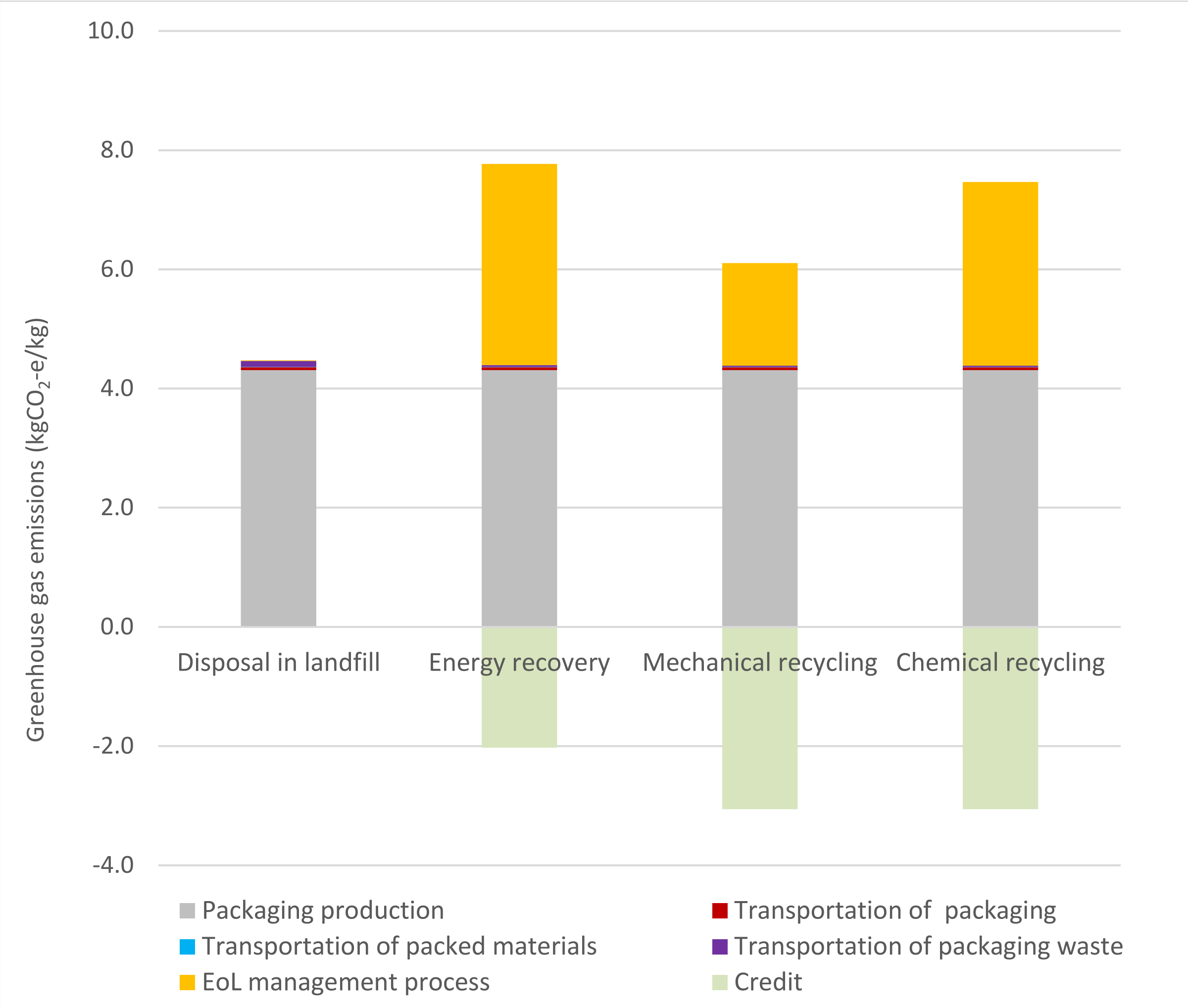

Figure 10. GHG emissions of construction LDPE packaging across four EoL management scenarios by activity
Source: Authors

Among three alternatives to disposal in a landfill, the mechanical recycling scenario exhibits the lowest total GHG emissions, at 6.10 kgCO_2-e/kg. This is primarily due to the relatively low energy requirements of the mechanical recycling process, making it the most favourable option. However, its effectiveness can be significantly reduced if the LDPE is contaminated or mixed with other waste, requiring additional washing or sorting steps (Ragaert et al., 2017).

The chemical recycling scenario has the second-lowest GHG emissions at 7.46 kgCO_2-e/kg. While this is about 22.3% higher than mechanical recycling, it may be a viable alternative for mixed or contaminated plastics that are unsuitable for mechanical recycling.

The waste-to-energy scenario has the highest GHG emissions at 7.77 kgCO_2-e/kg among the three alternatives. This is primarily driven by the conversion of the polymer to CO_2 during incineration, which releases its embedded carbon as CO_2 emissions, along with other GHGs.

The disposal in landfill scenario generates 4.47 kgCO_2-e/kg. This corresponds to approximately 4,137.79 kgCO_2-e/m³, using the average density of LDPE (925 kg/m³) (PlasticsEurope, 2025). This volumetric perspective is particularly relevant as waste management companies typically charge for collection based on volume ($/m³), while disposal costs and EPA landfill levies are mass-based ($/t). This creates an economic incentive structure where skip bin companies may preferentially recycle denser materials with established resale markets (like concrete and steel) while directing lighter materials like LDPE to landfill.

Although the three alternatives to disposal in landfill result in an overall increase in GHG emissions, landfilling itself comes with a significant cost. In the landfill disposal scenario, the entire GHG associated with the polymer's production—estimated at 3.60 kgCO_2-e/kg—is permanently lost as the material is removed from circulation. In contrast, mechanical and chemical recycling scenarios present an opportunity to offset much of the GHG emissions associated with polymer production through avoided virgin polymer production. At an 85% recycling efficiency, these scenarios can achieve an estimated savings of 3.06 kgCO_2-e/kg, representing a significant offset against their GHG emissions.

However, the market for recycled polymer in Australia is still developing (Shooshtarian et al., 2022). In the 2018–19 financial year, locally processed recycled polymers accounted for only 4% of the national consumption (O'Farrell, 2020). Recycled polymers are not usually preferred over virgin ones due to their higher cost of production and unknown quality. They often face higher costs due to factors such as labour, transport and infrastructure (Ghafoor et al., 2024). Their quality is also subject to debate with some standards prohibiting their use in certain applications (Santos et al., 2024). While mechanical recycling processes may degrade polymer quality over time, research suggests that polymers could be extruded up to 40 times without significantly altering their processability and long-term mechanical properties (Jin et al., 2012). Meanwhile, chemical recycling offers the potential to restore polymers to their original quality; however, practical limitations such as process efficiency and material loss prevent infinite recyclability (Achilias et al., 2007).

The waste-to-energy scenario also provides a credit for avoided electricity generation, offsetting emissions that would otherwise come from the current Victorian electricity grid, which relies on a mix of coal and natural gas. This credit, amounting to 2.02 kgCO_2-e/kg, offsets about 26% of the total scenario's GHG emissions. That said, the long-term viability of waste-to-energy is sensitive to the decarbonisation of national electricity grids. As grids increasingly rely on renewable energy sources, the relative carbon benefit of waste-to-energy will diminish, and its emissions profile will become less competitive compared to other EoL management options.

When looking at the activities within each scenario, packaging production — accounting for 4.31 kgCO_2-e/kg— is the dominant contributor. This activity alone accounts for more than 50% of total emissions across four scenarios. This is because LDPE as a fossil-based material has a high GHG coefficient and emphasises the importance of reducing emissions in LDPE production.

Transportation, on the other hand, was found to contribute a relatively small share of total GHG emissions, approximately 3.4% in the disposal in landfill scenario and 1% in the other three alternatives to landfilling. The difference is mainly due to differences in travel distance and the type of truck used for transportation. This relatively low contribution might, to some extent, be attributed to the assumption that all activities are locally sourced. This finding, however, aligns with previous studies that found transportation's share of GHG emissions to be low (Tan et al., 2023).

4.1 Limitations

This study faces several limitations. First, the system boundaries defined in the modelled scenarios may not encompass all relevant input and output processes. The assumptions made could also have influenced the results. While the study focused solely on GHG emissions, it is important to note that the life cycle of a product can also impact other critical areas, such as resource depletion. Furthermore, the inventory analysis was limited by the quality of data from the LCI database and the literature, though the data quality was deemed sufficient for the purpose of this study. Future research should consider the sensitivity of results to variations in the source of the polymer (virgin *vs* recycled), transport distances and modes, recycling efficiency rates and grid decarbonisation scenarios.

5. Concluding Remarks

LDPE is widely used in the construction industry for packing construction materials, but its short lifespan contributes to a significant waste stream, much of which is traditionally sent to landfill. This practice not only squanders the resources embedded in the production of LDPE but also poses long-term environmental and health risks due to its lengthy decomposition process. This study utilised LCA to compare the environmental impacts of construction LDPE packaging across four EoL management scenarios, namely, disposal in landfill (business-as-usual), waste-to-energy, mechanical recycling and chemical recycling. The analysis revealed that while disposal in landfills has low GHG emissions during the EoL stage itself, it permanently loses the embedded resources in LDPE production. Considering the offset achieved by the avoided virgin polymer production, mechanical recycling is the most environmentally favourable option. However, its effectiveness may decrease when dealing with mixed or contaminated waste. In such cases, chemical recycling, despite its higher GHG emissions, presents a viable alternative. Waste-to-energy, while useful in reducing waste volume, showed the highest GHG emissions, particularly as the decarbonisation of electricity grids reduces the relative benefit of its energy recovery. Among all activities, the production of LDPE packaging accounts for over 50% of the total GHG emissions across four scenarios, while transportation contributes a relatively low share, representing 1–3%.

These findings provide the following recommendations for practitioners and policymakers:

- Minimise the use of LDPE packaging in construction.
- Prioritise upstream strategies such as reducing the reliance on fossil-based and virgin polymers to reduce production GHG emissions.
- Separate LDPE packaging waste on-site to improve the quality of recyclables and streamline downstream processing.
- Implementing a waste management plan before construction begins to coordinate waste flows and increase the likelihood of successful recycling.
- Establish local reverse-supply chains to keep transportation costs and share of emissions as low as possible.
- Consider shifting waste disposal pricing and EPA levies for lightweight waste materials from weight-based ($/t) to volume-based ($/m³) to encourage their recovery.
- Maximise EoL recycling with a preference for mechanical recycling wherever feasible.
- Minimise the EoL management via waste-to-energy as it may not lead to any savings with the decarbonisation of electricity grids.

- Improve the market for recycled polymers through quality assurance, incentives and cost-efficiency measures.
- Foster stakeholder collaboration to align interests and encourage better design and EoL management.

By implementing these recommendations, a more sustainable life cycle for LDPE packaging can be achieved, helping to mitigate its environmental impact within the construction industry.

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