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White spill: Life cycle assessment approach to managing marine EPS litter from flood-released pontoons

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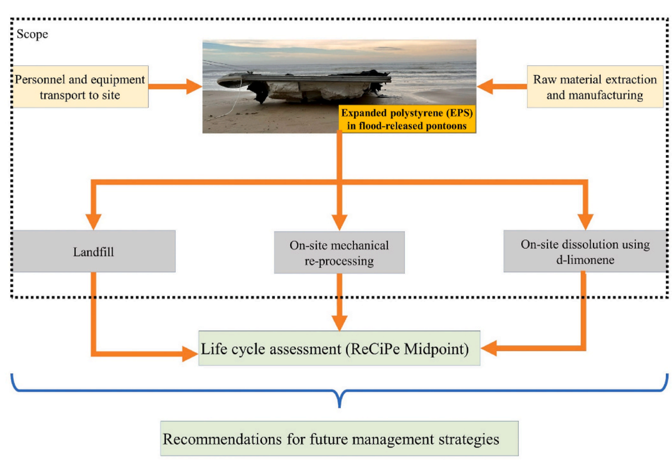
HIGHLIGHTS

- Transportation imposed the highest impacts during disposal of missing pontoons.
- Mechanical re-processing had highest impacts.
- D-limonene dissolution showed the best environmental performance.
- Further research is needed to optimise D-limonene dissolution process.
- Recovery of PS is desirable, but not feasible.

GRAPHICAL ABSTRACT



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ABSTRACT

Expanded polystyrene (EPS) pollution in the marine environment is a pressing issue in Queensland, Australia due to a recent flood that scattered hundreds of EPS-containing pontoons along the coastline, causing severe ecological damage. To assist in the clean-up effort and provide crucial data for developing management guidelines, this study investigates the environmental performance of different end-of-life (EoL) disposal/recycling methods, including (i) landfill; (ii) on-site mechanical re-processing using a thermal densifier (MR); and (iii) on-site dissolution/precipitation using D-limonene (DP). Applying the life cycle assessment framework, the results showed that DP was the most environmentally favourable option. Its impacts in climate change (GWP), acidification (TAP), and fossil fuel depletion (FFD) were 612 kg CO_2 eq, 4.3 kg SO_2 eq, and 184.7 kg oil eq,

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respectively. For comparison, the impacts of landfilling EPS in these categories were found to be 700 kg CO_2 eq, 3.5 kg SO_2 eq, and 282 kg oil eq, respectively. Landfill also contributed considerably to eutrophication potential (MEP), at 3.77 kg N eq. Impacts from MR were most significant due to the need to transport the densifier unit to the site. The analysis also revealed that the transportation of personnel and heavy machinery to the site, was the biggest contributor to impacts in the EoL stage. Its impacts in GWP, TAP, MEP, and FFD were 1369.8 kg CO_2 eq, 6.5 kg SO_2 eq, 0.2189 kg N eq, and 497.7 kg oil eq, respectively. Monte Carlo analysis showed that the conclusions made from these results were stable and reliable. Limitations of this model and recommendations for future investigations were also discussed in this work.

1. Introduction

Expanded polystyrene (EPS) is a common material used in aquatic and marina structures, such as pontoons, buoys, and floats. Its prevalence in this industry largely stems from its cost-effectiveness, high buoyancy, low density (15–30 kg/m^3), and low water adsorption (Fauna and Flora International, 2020). Unfortunately, this prevalence majorly contributes to an overabundance of EPS on our beaches and oceans (Fauna and Flora International, 2020; Lee et al., 2015). In many beaches around the world, such as the beaches of Salish Sea (Washington, US) and the East Sea (Korea), EPS is among the most common types of marine litter (Davis and Murphy, 2015; Hong et al., 2014). The abundance of EPS fragments in the marine environment poses a serious ecological concern as seabirds, fish, turtles, turtle hatchlings, and other faunas can mistake EPS beads for food, which can cause internal injuries or death (Battisti, 2020; Campani et al., 2013; Egbeocha et al., 2018). Additionally, the breakdown of EPS presents a source of microplastics and additives, which can bioaccumulate through the food web (Li et al., 2019). This risk is further exacerbated by the conditions of the beach, which enhances the rate of EPS fragmentation due to increased exposure to wind and sand abrasion, high sand temperature, and strong UV ray from sunlight (Song et al., 2017; Turner, 2020).

The issue of EPS pollution in the marine environment is particularly pressing in Queensland, Australia, where a serious flooding event scattered hundreds of large pontoons, each containing up to 600 kg EPS, along the coastline. The resulting EPS pollution was dubbed the "white spill" (Mapstone et al., 2022), and its implication to the local marine life can be dire, especially when there is no sustainable measure to efficiently manage EPS waste (Figure S1). This incident highlighted the urgent need to develop a suitable strategy to protect the marine environment and minimise the potential damage of EPS originated from pontoons in future floods. The main challenges with managing flood-released pontoons are primarily associated with the low bulk density of EPS, which massively increases the total volume of waste that needs to be disposed. This enormity means EPS-containing pontoons can occupy large spaces in the local landfill and shorten its lifespan. Furthermore, transporting and processing EPS pontoons can also incur significant costs, logistical considerations, and environmental impacts due to their low weight/volume ratio (Mumbach et al., 2020). This is especially true when pontoons are found in remote areas and must be transported long distance to an appropriate treatment facility. Additionally, EPS is highly brittle and can fragment into small particles during transportation and handling, which further increases the risk of releasing plastics into the environment (Marten and Hicks, 2018). Considering these risks, life cycle assessment (LCA) frameworks can be utilised to support evidence-based decision-making and guide research directions. Considering these risks, life cycle assessment (LCA) frameworks can be utilised to support evidence-based decision-making and guide research directions.

Few LCA studies have been conducted to assess the environmental impacts associated with EPS products and/or their end-of-life options. Lim et al. (2021) assessed the environmental merits of treating EPS waste using landfill, incineration, and mechanical recycling. The analysis found that incinerating EPS waste had the highest environmental impact, whereas recycling had the least. EPS reuse and recycling also exhibited the most environmental benefits in a study by Lindstrom and Hicks (2022), who conducted LCA for EPS shipping boxes. Contrary, in another LCA study, Tan and Khoo (2005) compared the environmental impacts of disposing EPS packaging materials using landfills and incinerators and found that the latter option was more favourable. Another study by PWC (2011) focused on the cradle-to-grave impacts of fishbox packaging systems made from EPS. The report noted that the production of raw materials accounted for 40–60% of the total energy consumption, greenhouse gas emissions, and acidification potential of EPS. It should be noted that these LCA models were developed for the management of packaging materials in Singapore (Tan and Khoo, 2005), Malaysia (Lim et al., 2021), the USA (Lindstrom and Hicks, 2022), France, Spain, and Denmark (PWC, 2011). Thus, findings from these studies have limited transferability to the management of derelict EPS pontoons in Queensland, Australia. Additionally, these studies did not explicitly address the additional fuel consumption during transportation and landfill burden due to the bulky nature of EPS materials. For example, many studies calculated the landfill infrastructure burden based on methodologies developed by Doka (2003), which assumed the waste to have a density of 1000 kg/m^3. Though this method worked well for denser waste streams such as organic waste or hard plastics, it may be inaccurate for bulky EPS pontoons. Thus, LCA studies for EPS products may have underestimated the environmental impacts of managing this waste. This lack of information poses significant challenges in developing an evidence-based management strategy to protect our coastlines from "white spills".

Considering the gaps presented, the current study is dedicated to developing cradle-to-grave LCA models to quantify the environmental impacts/benefits of different disposal/recycling methods for EPS-based pontoons found along Queensland beaches. Particularly, this work developed a case-specific LCA model to quantify the impacts of treating EPS-based pontoons via (i) landfill (LF); (ii) on-site mechanical reprocessing using a thermal densifier (MR); and (iii) on-site dissolution/precipitation (DP) using D-limonene as the solvent. The novelty of this work stems from special considerations for the increased environmental effects associated with transporting and disposing high-volume, low-density EPS pontoons. The environmental impacts were evaluated using multiple impact categories, such as climate change potential, acidification potential, and eutrophication potential. Findings from this work can help identify and minimise environmental "hotspots" in the management of flood-released pontoons. Additionally, the models developed in this paper can help optimise the logistics associated with collecting and disposing of derelict pontoons, which can reduce delays and decrease the risk of EPS leakage into the environment. Lastly, recommendations and conclusions derived from this work can be incorporated as part of crisis-response strategies/guidelines to minimise and/or prevent the environmental damages of "white spill" should similar urgencies occur in the future.

2. Methodology

The LCA modelling in this study was conducted using standardised criteria and methodologies, particularly those established and maintained by the International Organisation of Standardisation (ISO). These standards included ISO 14044:2006 (ISO, 2006) and ISO 15270:2008

(ISO, 2008). Additionally, recommendations by Keller et al. (2022) for increased transparency, comprehensiveness, and comparability of LCA for chemical recycling processes were considered when analysing dissolution/precipitation treatment method. SimaPro version 8.2.0, published by PRé Sustainability, 2023 was used to (i) generate the LCA results; and (ii) test the sensitivity of the results by applying integrated Monte Carlo analysis. These steps ensured that the results and recommendations provided by this work are reliable and re-producible.

2.1. Goal and scope

The goal of this study is to use LCA methodology to holistically quantify the life cycle impacts of management scenarios for flood-released pontoons found along the beaches of Queensland following flooding events. These scenarios are described in Table 1 and further elaborated in the subsequent sections. Although the focal point of this study is the end-of-life (EoL) treatment of EPS waste in pontoons, the system boundary developed for this model included all life cycle stages of EPS, such as material extraction, manufacturing, and EoL management (Fig. 1). The EoL management phase was further broken down to include (i) transportation of personnel and heavy machineries to/from the pontoons; (ii) transportation of EPS to a landfill; (iii) the use of heavy machineries to handle the pontoon and/or separate EPS components from the pontoons; (iv) transportation of on-site treatment equipment (thermal densifier or D-limonene tank) to/from the sites; (v) D-limonene production; and (vi) final EoL disposal/recycling. The transportation and disposal of the inert components of the pontoons, particularly steel and concrete, were excluded from this analysis. Details regarding the assumptions and calculations made for each life cycle stages have been discussed and disclosed in Section 2.2.

The functional unit (FU) for this study is the treatment of 600 kg of EPS, which was the reported amount of EPS in a full-size pontoon found on Queensland beaches. To ensure appropriate representation of relevant environmental impacts and their main drivers while avoiding excessive complexity and data requirements, any material and energy flows contributing less than 1% of the total flow were omitted from the analysis.

2.2. Inventory analysis

2.2.1. Raw material extraction

The raw material acquisition phase of EPS consists of extractive and refinery activities to obtain oil and gas. The products of refinery are benzene, ethylene, and pentane, which are used in the manufacturing phase to produce EPS for pontoons. Data pertaining to unit processes involved in this stage was taken from ecoinvent 3.0 database (Fehrenbach et al., 2018; Jing et al., 2020; Meili et al., 2022; Althaus, 2007).

Table 1
End-of-Life (EoL) treatment scenarios of flood-released EPS-based pontoons.

Nomenclature	EoL scenario	Description
LF	Landfill	EPS is disposed of in a sanitary landfill without on-site size reduction.
MRne	Mechanical re-processing (without PS recovery)	EPS is mechanically re-processed using a thermal densifier. The densified ingot is landfilled.
MRwE	Mechanical recycling (with PS recovery)	EPS is mechanically re-processed using a thermal densifier. The densified EPS ingot is recycled (substitution factor: 0.5)
DPne	Dissolution/ precipitation (without PS recovery)	EPS is dissolved and re-precipitated into a denser material, which is disposed of in a landfill.
DPwE	Dissolution/ precipitation (with PS recovery)	After re-precipitation, the material is reused to manufacture pure PS (substitution factor: 0.95)

2.2.2. Manufacturing

The manufacturing of EPS starts by combining benzene and ethylene to produce styrene. Styrene is polymerised with the help of a catalyst, the most common of which is organic peroxide. Polystyrene (PS) is transformed into EPS using steam and a minute amount of pentane, which expands the polystyrene up to 40 times its original volume (FOAMEX, 2021). Impacts associated with the distribution of EPS were also included in this phase. The quantity of benzene, ethylene, steam, and pentane needed to produce 600 kg of EPS, as well as emissions, energy consumption, and material usage data, were taken from ecoinvent 3.0 (Althaus, 2007) and Tan and Khoo (2005).

2.2.3. Transportation

The transportation stage is a crucial part in the management of flood-released pontoons, as it can significantly contribute to the overall environmental impacts of EoL treatment. Modelling transportation for this study is particularly complex for several reasons. Firstly, these pontoons can be massive (Figure S2), weighing up to 15 tonnes and having average dimensions of 14 m × 4 m × 1 m (total volume of 56 m^3). Thus, Oversize Overmass Vehicles (OSOM) are needed to transported large equipment to the sites (excavators, loaders, cranes). Secondly, there were more than 300 identified pontoons carried to open waters by the floods and had to be towed back to port. This can be particularly energy-intensive and challenging for hauling trucks and heavy machinery to access. Lastly, a fleet of utility vehicles is needed to transport personnel to/from the site, which further adds to the burden in managing this waste.

This study assumed that a 28t truck is used to transport all the heavy equipment, which had a total weight of 40 tonnes, to/from the site. Transportation distance was assumed to be 150 km each way. Thus, the total energy spent on transporting this equipment was calculated as 12,000 t·km, where one t·km is defined as the energy required to move one tonne of goods over 1 km. A separate 12t truck-mounted crane and a tractor are also needed. Their travel distance to/from the site was assumed to be 150 and 10 km, respectively. Furthermore, passenger cars, each travelling 150 km, were assumed to have equivalent impact to the utility cars. Fuel consumption and emission data associated with the operation and maintenance of these vehicles were based on publications by Australian Transport Assessment and Planning (2016), Truck Impact Chart (Australian Truck Association, 2018), and the Australian Transport Facts (Adam Pekol Consulting, 2011). Life cycle inventory (LCI) of indirect impacts and background processes were populated using the ecoinvent database (Notten et al., 2018; Spielmann and Scholz, 2005).

2.2.4. Use of heavy machineries

The enormity of the pontoons found on the beaches of Queensland necessitates the use of heavy industrial machines, such as cranes, excavators, loaders, and tractors. These pieces of equipment are needed to handle and load the pontoons onto the transport truck. Moreover, for MR and DP scenarios, additional use of these machineries is needed to help separate EPS components from the rest of the pontoon. Data pertaining to diesel consumption, air emission, maintenance, and lubricant use from this stage was primarily taken from the ecoinvent database 3.0 (Frischknecht, 2007; Kellenberger and Althaus, 2007), AusLCI (Life Cycle Strategies, 2015), and relevant manufacturers. This data is presented in Table 2, along with the assumed operating hours.

2.2.5. Sanitary landfill

The first end-of-life treatment scenario being modelled was sanitary landfill disposal (LF). In this scenario, it was assumed that the beached pontoon was transported to a local landfill without on-site size reduction (Eumundi-Noosa Rd landfill). A 28t truck was used to transport the pontoon to the landfill. The total transportation distance was assumed to be 300 km. It should be noted that only impacts of transporting 600 kg EPS was calculated. Impacts associated with transporting the inert component were omitted despite being transported along with the EPS.

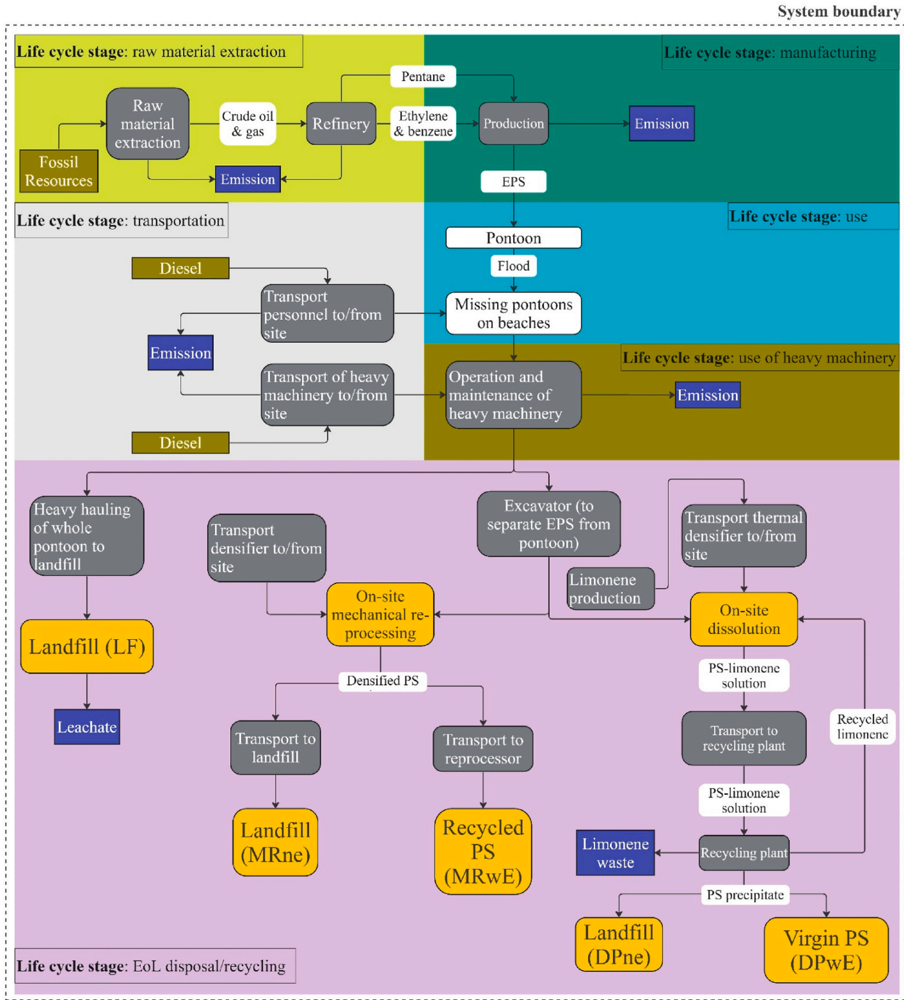

Fig. 1. System boundary and end-of-life (EoL) treatment scenarios considered for this work.

Table 2
Transportation stages and associated assumptions (per functional unit).

Equipment	Operating input	Reference
Crane	3 h	(Tadano, 2020; Tadano, 2014; Palfinger Sany, 2016)
Hydraulic excavator	10 h	(Hitachi, 2015; Kellenberger and Althaus, 2007)
Skid steer loader	10 m^3	Kellenberger and Althaus (2007)
Front-end loader	8 h	(JCB, 2013; JCB, 2017)
90–120 hp tractor	1 t km	Lattz and Schnitkey (2017)

Thus, the freight load was calculated as 180 t·km.

The destination landfill had a total capacity of 2,753,000 m^3 (Ferris and Florence, 2016). Based on a density of 15 kg/m^3 (Engineering ToolBox, 2009), the volume of 600 kg EPS was calculated as 40 m^3. Thus, landfill infrastructure burden of disposing of EPS was calculated as 1.45×10^{-5} p^{-1}. Landfill gas emission from pontoon was assumed to be negligible over 100 years, which was the time horizon applied to this study. This is because EPS, being a type of plastic, has an extremely low rate of degradation, as demonstrated in Bigger et al. (2012) and Xochitl (2021). This study assumed that 1.92 m^3 of leachate was produced from 600 kg EPS in 100 years, based on the estimation from Yang et al. (2015). The primary pollutant of concern in the leachate is microplastic, which is assumed to be present at a concentration of 2.72 mg/L (Narevski et al., 2021). Using this information and the methodology developed by Doka (2003), the total burden from leachate treatment was calculated as 3.84×10^{-5} p^{-1}. Background data concerning electricity and diesel consumed during landfill operation was taken from ecoinvent (Doka, 2003).

2.2.6. *Mechanical Re-processing*

The mechanical re-processing technology modelled in this paper was based on the GreenMax Foam Densifier *M*-C300, manufactured by Intco Recycling (2012a). The thermal-based densifier is a specialised machine designed for compacting EPS waste with a compression ratio of 90:1. This equipment was chosen for its portability, the availability of an Australian supplier, and its prior use by Queensland Logistics Service (Intco Recycling Australia, 2023; Intco Recycling Australia, 2023). It had a capacity of 300 kg EPS/h and dimensions of 3.8 m × 4.48 m × 4.4 m (L × W × H) and weighs 3.2 tonnes (Intco Recycling, 2012b). Thus, a 6t truck was assumed to be used to transport the densifier to the beach for on-site treatment (distance: 300 km). As the volume of the original EPS had been significantly reduced, only a light commercial vehicle (LCV) was used to transport the densified ingot back to an appropriate EoL facility. The densifier used a total of 113.7 kW h, based on 3-h operations (Intco Recycling, 2012a). Electricity was provided by an on-site diesel generator. This study assumed that an excavator with an operation time of 5 h was used to help separate EPS from the rest of the pontoon. LCI pertaining to the operation and maintenance of the 6t truck, LCV, diesel generator, and excavator was taken from the ecoinvent database (Kellenberger and Althaus, 2007; Notten et al., 2018).

Two EoL scenarios were derived from the use of the Foam Densifier

M-C300 equipment (Fig. 1). The first scenario assumed that the densified PS ingots, which had a density of 700 kg/m^3, were transported and discarded at the Eumundi-Noosa Rd landfill (MRne). Thus, the landfill infrastructure burden of the PS ingots was calculated to be 3.113×10^{-7} p^{-1}. The total burden from leachate treatment and transportation distance were calculated the same way as that of the landfill scenario. The second scenario assumed that the product from MR was reused as a source of PS pellets to manufacture recycled products (MRwE). To account for "downcycling' effects due to contamination and heat degradation, this study assumed that 2 kg of recovered PS displaced 1 kg of virgin-grade PS pellets (substitution factor: 0.5). Thus, credits for avoided impacts were given accordingly. This assumption was tested in Monte Carlo analysis.

2.2.7. Dissolution/precipitation using D-limonene

Treating EPS waste using the DP method is particularly interesting because it represents an emerging waste management option that can increase the efficiency of EPS waste management (Achilias et al., 2009). The generic framework of DP recycling involves dissolving the EPS in a solvent (benzene, toluene, acetone, *p*-cymene, D-limonene, etc.). The solvent is often highly selective, thus only EPS is dissolved while contaminants remain solid. The contaminants are filtered out, while the EPS-solvent mixture is separated by re-precipitating the PS. The PS precipitate is now 20 times denser than the original EPS and is comparable to virgin-grade PS pellets (Zhao et al., 2018; Noguchi et al., 1998a). It should be noted that there is limited public sentiment for the use of petrochemical solvents, such as benzene, toluene, and acetone, due to their inherent incompatibility with circular economy and high environmental toxicity. Thus, D-limonene (C10H16; CAS number: 5989-54-8) was chosen as the solvent for this study due to its natural origin (orange peel oil extract), and the availability of high-quality inventory data (Hattori, 2015; García et al., 2009a). Furthermore, this solvent can be used in ambient temperature, which is extremely important as on-site energy supply can be limited (Hattori, 2015; García et al., 2009a). Furthermore, this solvent can be used in ambient temperature, which is extremely important as on-site energy supply can be limited (Achilias et al., 2009).

Since there is no specific data available on the use of D-limonene to dissolve EPS in pontoons, the most prudent approach would be to rely on existing literature and reports from the industry. The recycling system developed for this scenario was similar to that proposed by Noguchi et al. (1998b, 1998c), with some modifications. Particularly, a tank containing 2000 L of limonene (95% purity) can be mounted on a truck and transported to the beach site (Figure S3). The EPS component was manually removed from the pontoon with the help of an excavator and placed in the tank to be dissolved. No heat was provided to the system. Once all the EPS was dissolved in the tank, the truck can transport the PS-limonene mixture, containing 26 wt% PS, to a recycling plant. At this plant, the solution was passed through a filter to remove contaminants. Thence, PS and D-limonene were separated using vacuum distillation (Achilias et al., 2009; Garcia et al., 2009b).

Impacts of D-limonene production were based on LCI published by Pourbafrani et al. (2013) and Teigiserova et al. (2022). Additionally, environmental fates and burden of D-limonene were taken from the National Industrial Chemicals Notification and Assessment Scheme (NICNAS, 2002). D-limonene was assumed to be produced in South-East Asia and transported to Australia via transoceanic ships. This investigation assumed that D-limonene can be reused for 10 cycles, based on work by Noguchi et al. (1998b). A 2% loss of limonene during transportation was assumed in this EoL scenario. Discarded D-limonene was treated as hazardous waste. Infrastructure burden of the recycling plant was calculated as 4.17×10^{-7} p^{-1}, based on a "typical" plant put forward by Althaus (2007). Impacts associated with the operation and maintenance of the truck transporting the D-limonene were assumed to be equivalent to that of an LCV (distance: 300 km).

Two scenarios were developed from this scheme. One scenario (DPne) involved the landfill disposal of the PS precipitates following extraction from the PS-limonene mixture. Previous works noted that D-limonene dissolution reduced the volume of EPS by at least 20 times (Noguchi et al., 1998b; Kan and Demirboga, 2009). Thus, the PS precipitate was calculated to have a density of 300 kg/m^3. Therefore, the landfill infrastructure burden for disposing of the EPS component of the pontoon was calculated to be 6.226×10^{-7} p^{-1}. The burden from leachate treatment and transportation to landfill was identical to that in the LF scenario. The other DP scenario involved using the PS precipitate as a source for virgin-grade PS pellets (DPwE). The DPwE scenario allocated credits for avoided PS production by conventional means. As many studies have noted that the recovered PS had identical molecular weight and physical properties as that of virgin PS pellets, this study used a substitution factor of 0.95 (Noguchi et al., 1998b; Achilias et al., 2009).

Background LCI data was taken from ecoinvent (Frischknecht et al., 2007) and AusLCI (2011). To address the data gap for this EoL alternative, the worst-case scenario data was considered. This means applying the most unfavourable material/energy consumption and emission rate found in the literature. For example, this would involve using the highest solvent consumption rate reported and maximum losses of D-limonene during transportation and on-site treatment were assumed. Furthermore, Monte Carlo analysis was applied to all the data to ensure that the LCA results are still valid despite data uncertainty. This is further explored in the following section.

2.3. Impact assessment and sensitivity analysis

The life cycle impacts of different EPS treatment scenarios were assessed using Recipe Midpoint (Hierarchist) methodology, with a time horizon of 100 years. The following impact categories were used: climate change potential (GWP, kg CO_2-eq), freshwater eutrophication (FEP, kg P-eq), marine eutrophication (MEP, kg N-eq), terrestrial acidification (TAP, kg SO_2-eq to air), and fossil fuel depletion (FFD, kg oil-eq). Details of the assessment method and description of impact categories can be found in Huijbregts, 2016.

This study also applied system expansion approach to MRwE and DPwE scenarios to account for the avoided burden resulting from the displacement of virgin PS material by recovered PS. Using this methodology, environmental benefits of recycling processes can be allocated and subtracted from the total environmental impact (Heijungs et al., 2021; Weidema, 2014). No weighting and normalisation were applied to the result.

Monte Carlo analysis was conducted in SimaPro version 8.2.0, with 1000 runs at 95% confidence interval. This analysis aimed to evaluate the sensitivity of the result to variations in the input parameters. This type of analysis is extremely useful as it provides important insight into how assumptions made during LCA modelling, such as operating hours of heavy machinery, transportation distances of trucks, substitution factors, and diesel consumptions, impacted the result.

3. Results

3.1. Overall impacts

The characterised impacts for all life cycle stages and EoL scenarios are presented in Table 3. The raw material extraction stage had the highest impacts in GWP and FFD, at 2028 kg CO_2 eq/FU and 1140 kg oil eq/FU, respectively. This stage also had considerable impacts in TAP, FEP and MEP, at 5.9 kg SO_2 eq/FU, 0.0651 kg P eq/FU, and 0.1340 kg N eq/FU, respectively. The manufacturing stage had the third-highest TAP, at 5.6 kg SO_2 eq/FU. Its impacts in GWP, FEP, MEP, and FFD were 618.3 kg CO_2 eq/FU, 0.0153 kg P eq/FU, 0.1133 kg N eq, and 174.9 kg oil eq/FU, respectively. In total, the production of 600 kg EPS, which included raw material extraction and manufacturing, emitted 2646 kg CO_2 eq and 11.5 kg SO_2 eq. This result was reasonably similar to

Table 3
Cradle-to-grave impact scores of expanded polystyrene found in flood-released pontoons, with particular attention given to end-of-life (EoL) management. MRne: mechanical re-processing without recovery; MRwE: mechanical re-processing followed by polystyrene recovery; DPne: dissolution/precipitation without recovery; DPwE: dissolution/precipitation followed by polystyrene recovery. Negative values represent environmental benefits from avoided PS production.

Impact categories	Raw material extraction	Manufacturing	Transport	Use of heavy machinery	EoL Scenarios				
					LF	MRne	MRwE	DPne	DPwE
GWP (kg CO_2 eq/FU)	2028.1	618.3	1369.8	591.8	699.9	1132.0	580.2	612.0	−414.5
TAP (kg SO_2 eq/FU)	5.9	5.6	6.5	2.9	3.5	5.5	−8.8	4.3	−22.9
FEP (kg P eq/FU)	0.0651	0.0153	0.0516	0.0176	0.0243	0.0368	0.0368	0.0955	0.0910
MEP (kg N eq/FU)	0.1340	0.1133	0.2189	0.1576	3.7672	1.9039	−0.0345	1.2981	−0.3959
FFD (kg oil eq/FU)	1140.0	174.9	497.7	204.1	282.0	375.5	61.1	184.7	−421.4

that of Lim et al. (2021), who reported 2990 kg CO_2 eq and 7.38 kg SO_2 eq were emitted for every tonne of EPS produced.

In terms of life cycle stages specific to the premise of this work, the transportation stage had the highest impacts in GWP, TAP, and FFD, at 1369.8 kg CO_2 eq/FU, 6.5 kg SO_2 eq/FU, and 497.7 kg oil eq/FU, respectively. Life cycle impact of transportation in FEP and MEP were moderate at 0.0516 kg P eq/FU and 0.2189 kg N eq/FU, respectively. The use of heavy machinery had intermediate impacts, the most significant of which were GWP, TAP, and FFD, at 591.8 kg CO_2 eq/FU, 2.9 kg SO_2 eq/FU, and 204.1 kg oil eq/FU, respectively.

Across all impact categories, the EoL scenarios ranking order (including uncertainty) was: DPwE > MRwE > DPne > LF > MRne. DPwE had the best environmental performance amongst all EoL scenarios, providing considerable quantified benefits across nearly all impact categories. Its net impact in GWP, TAP, MEP, and FFD were −414.5 kg CO_2 eq/FU, −22.9 kg SO_2 eq/FU, −0.4 kg N eq/FU, and −421.4 kg oil eq/FU, respectively. However, this scenario had the second-highest impact in FEP, at 0.091 kg P eq/FU. MRwE offered similar benefits, but to a lesser extent. Its impacts in GWP, TAP, MEP, and FFD were 580.2 kg CO_2 eq/FU, −8.8 kg SO2 eq/FU, −0.0345 kg SO_2 eq/FU, and 61.1 kg oil eq/FU, respectively. However, the FEP impact of this scenario was significantly lower than that of DPwE, at 0.0368 kg P eq/FU. DPne had the highest impact in FEP, at 0.0955 kg P eq/FU. Its impacts in GWP, TAP, and FFD were 612 kg CO_2 eq/FU, 4.3 kg SO_2 eq/FU, and 184.7 kg oil eq/FU, respectively. In other words, for every kg of EPS treated, 1.02 kg CO_2 eq and 7.2×10^{-3} kg SO_2 eq were emitted, and 0.3 kg oil eq was consumed. For comparison, Noguchi et al. (1998c) reported 0.79 kg CO_2 and 4.2×10^{-3} kg SO_2 emission, and 0.28 kg oil consumption, which validated our results.

LF imposed the highest impact in MEP across all life cycle stages, at 3.77 kg N eq/FU. Furthermore, it also had relatively high impacts in GWP, TAP, and FFD, at 699.9 kg CO_2 eq/FU, 3.5 kg SO_2 eq/FU, and 282 kg oil eq/FU, respectively. Amongst all EoL scenarios, MRne had the highest impact in GWP, TAP, and FFD, at 1132 kg CO_2 eq/FU, 5.5 kg SO_2 eq/FU, and 375.5 kg oil eq/FU. Details pertaining to the main process drivers of these results are presented in the following sections.

3.2. Process contributions & impact drivers

Contributions from different types of transportation and heavy machinery are presented in Fig. 2. Additionally, the characterised impacts and highlights of processes that contributed to the total impact of each EoL scenario are presented in Figs. 3–7, at 1% cut-off. The error bars represent the potential ranges of the impacts score, which were a function of uncertainty in input parameters. Diamond symbols represent the mean impacts calculated from Monte Carlo analysis. As this LCA investigation employed a system expansion approach, results with negative values can be interpreted as "avoided impacts" or beneficial effects on the environment. Conversely, positive values represent adverse impacts. "Remaining processes" in the legend referred to processes that were below the 1% cut-off threshold. The unit processes reported here are presented in percent of the total environmental impact (i.e., the net impact totals 100%). For example, if the sum of the burden

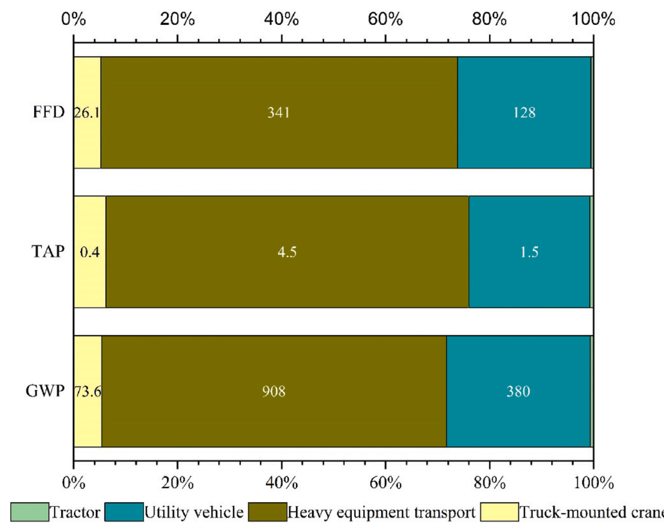

Fig. 2. Process contribution for transportation stage. x-axis showed percent contribution, values inside the bar represent absolute impact in GWP (unit: kg CO_2 eq/FU), TAP (unit: kg SO_2 eq/FU), and FFD (kg oil eq/FU). The transportation of heavy equipment to the site accounted for most of the impacts in GWP, TAP, and FFD, followed by personnel transportation using utility cars. NOTE: impacts in FEP and MEP were not shown as they were deemed insignificant.

is 150%, the sum of the credits for avoided impact must be −50% to achieve a net impact of 100%.

During transportation, impacts associated with moving heavy equipment to/from the site were the biggest contributors to GWP, TAP, and FFD, accounted for 66.3, 69.6, and 68.6% of the total impact of the transportation stage (Fig. 2). This was followed by impacts from the operation of utility vehicles, contributed 29.3, 25, and 27.1% to GWP, TAP, and FFD, respectively (Fig. 2). Impacts arising from transporting truck-mounted cranes and tractors to/from the site were found to be insignificant. Impacts during heavy machinery use are presented in Figure S4, with excavator operations accounted for 60.0% of impacts in both GWP and FFD. The operation and maintenance of backhoe loaders were the main driver for TAP, represented 49.3% of the impact. For both transportation and machinery use stages, the primary driver of impacts was due to fuel consumption and airborne emissions (>95%).

GWP is one of the most important categories in LCA as it indicates greenhouse gas (GHG) emission of a process. For the LF scenario, fuel use during the operation of the 28t truck to transport EPS to a landfill was the primary impact driver, contributed 63.4% of the total impact (Fig. 3). This finding was similar to that reported by Noguchi et al. (1998c), who noted that EPS collection accounted for nearly 70% of GHG emission during EoL stage. Landfill infrastructure burden accounted for 6.63% of the total impacts. For MRne, the primary contributors were fuel consumption in operating the 6t truck, LCV, and excavator, accounted for 42.4, 11.1, and 11% of the total impact, respectively. MRwE had similar GWP impacts, though the percentage

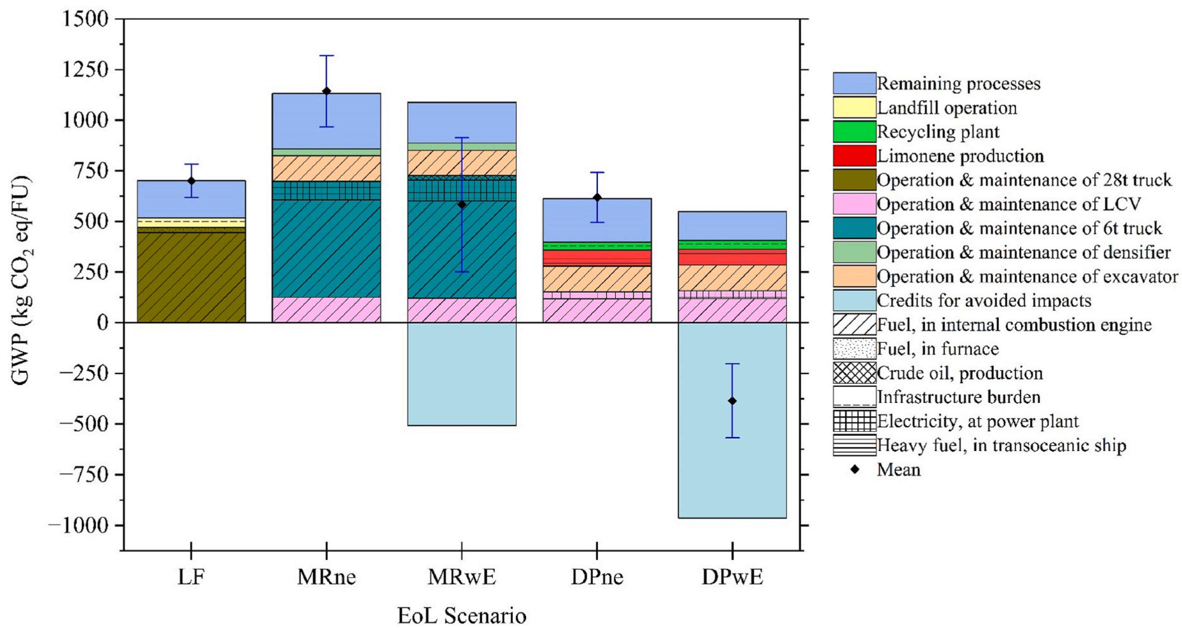

Fig. 3. Process contribution for climate change potential at 1% cut-off. The black diamonds indicate the mean net impacts from Monte Carlo analysis, with a positive number representing an adverse environmental impact and a negative number representing environmental benefits. This result showed that GHG emission from transporting the thermal densifier unit to the site in MR contributed to its greater impact LF scenario. GWP impact was the lowest during DP treatment.

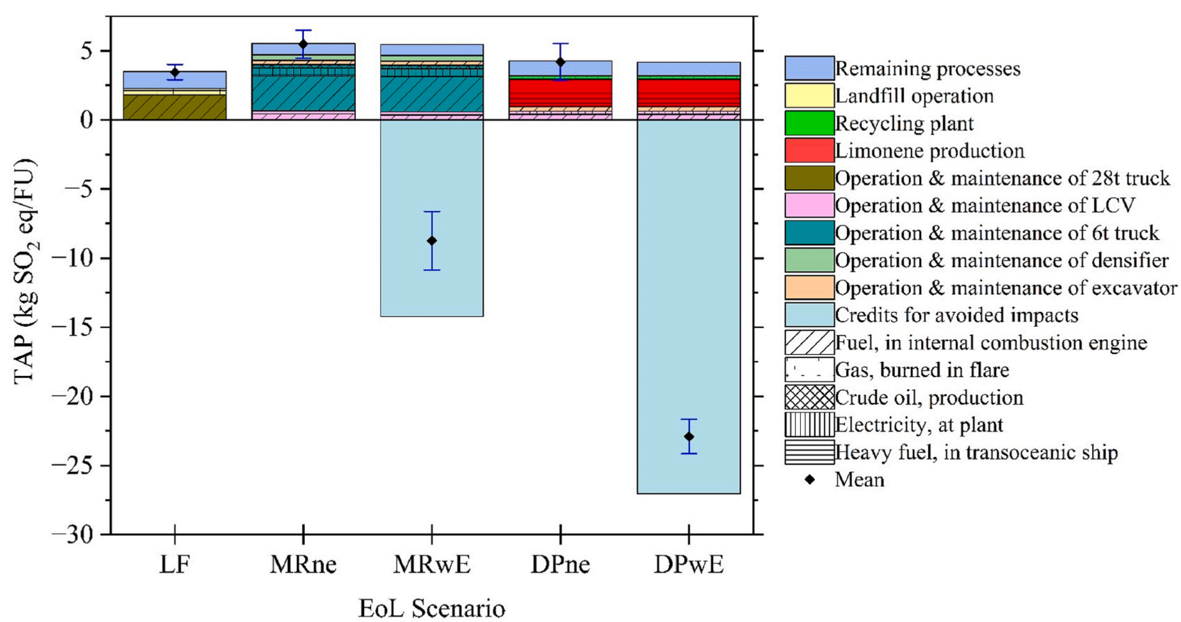

Fig. 4. Process contribution for terrestrial acidification potential at 1% cut-off. The black diamonds indicate the mean net impacts from Monte Carlo analysis, with a positive number representing an adverse environmental impact and a negative number representing environmental benefits. This result showed tremendous savings in SO_2 emissions due to avoided impacts from recovered PS in MrwE and DpwE scenarios. Use of vehicles contributed heavily to TAP impacts in all scenarios.

contributions were 44.2, 11.1, and 11.5% of the total impacts, respectively. The plastic credits for general-purpose PS pellets represented −46.6% of impacts. For DPne, the primary drivers of impacts were fuel use in LCV, excavator, and transoceanic ship (during D-limonene production). These impacts represented 19.2, 20.5, and 13% of the total GWP, respectively. For DPwE, the primary impact drivers were fuel use in LCV, excavator, and transoceanic ship, accounted for 21.6, 22.9, and 14.5% of impacts, respectively. Credits for recovered PS represented −175.5% of the total impacts in GWP.

TAP is an important impact category in assessing sulphur dioxide emission. For LF, the primary contributor to TAP was the operation of the 28t truck, accounted for 51.6% of the total impact (Fig. 4). For MRne, the main drivers were fuel and electricity consumption in operating and maintaining the 6t truck, represented 45.8 and 10.6% of the total impacts, respectively. MRwE had similar drivers, accounting for 46.5 and 11% of the impacts, respectively. Credits for avoided PS production represented a negative contribution of −261% to the total impact. For DPne, the primary contributors to TAP were fuel use in transporting D-limonene and operating LCV, contributed 46.9 and 9.5% of impact, respectively. DPwE also had similar primary impact drivers, with fuel use in transoceanic ship and operating LCV accounted for 48 and 9.8% of total impacts, respectively. Benefits gained from recovery of

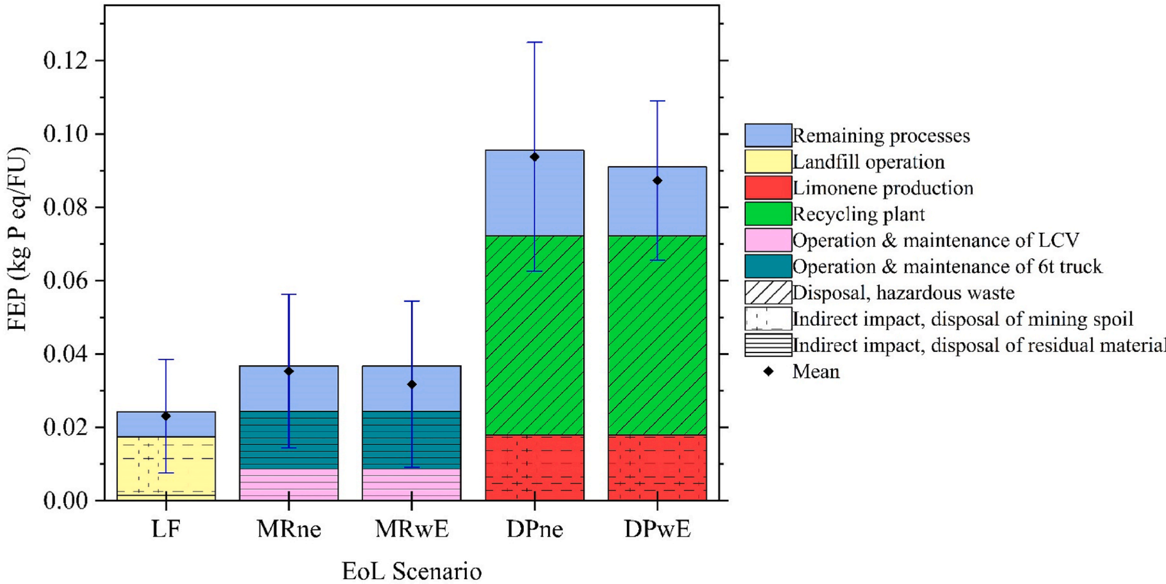

Fig. 5. Process contribution for freshwater eutrophication potential at 1% cut-off. The black diamonds indicate the mean net impacts from Monte Carlo analysis, with a positive number representing an adverse environmental impact and a negative number representing environmental benefits. This result showed that the disposal of D-limonene waste as a hazardous material was the main contributor to FEP impacts in DP scenarios.

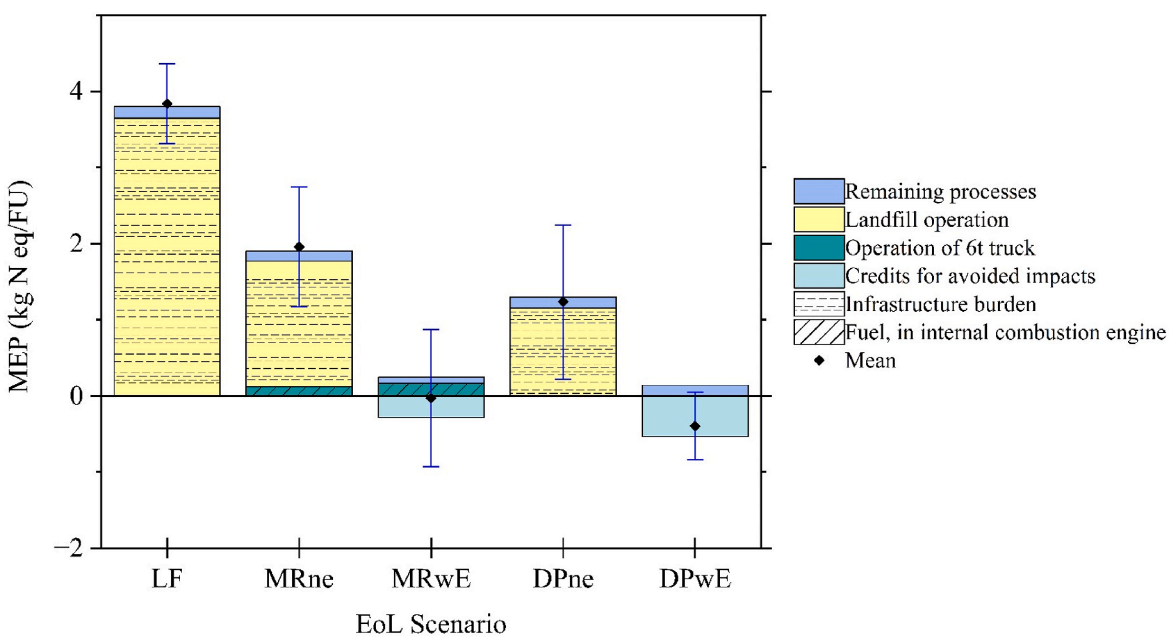

Fig. 6. Process contribution for marine eutrophication potential at 1% cut-off. The black diamonds indicate the mean net impacts from Monte Carlo analysis, with a positive number representing an adverse environmental impact and a negative number representing environmental benefits. The graph indicated that infrastructure burden from landfill, particularly due to leachate production, was the main driver of impact in MEP.

PS represented a negative contribution of −647% of impacts in this scenario.

In FEP category, the primary contributor for LF was due to indirect impacts from the disposal of mining spoils, representing 65.8% of impacts (Fig. 5). This impact was associated with coal mining activities to generate energy for landfill operations. For both MRne and MRwE, the primary process contribution was the indirect impacts from the disposal of residual materials associated with the operation and maintenance of the 6t truck and LCA, accounted for 42.6 and 23.7% of impacts, respectively. For DPne and DPwE, the drivers were indirect impacts from residual material disposal associated with recycling plant infrastructure burden (56.9 and 59.7% of impact, respectively), followed by indirect impact from spoil management during D-limonene production (18.8 and 19.7% of impacts, respectively).

For the MEP impact category, the primary impacts stemmed from P emission linked to landfill infrastructure burden. Thus, in LF, MRne, and DPne, this process contributed 96, 87, and 90% of the total impacts. Impacts from other processes were insignificant by comparison. Benefits gained from recovered PS in terms of MEP in MRwE and DPwE scenarios represented −379 and −114% of total impacts, respectively (Fig. 6).

FFD indicates the impacts of EoL scenarios on our fossil fuel resources. For LF, the primary contributors were crude oil production for the 28t truck and petroleum production associated with building and operating the landfill, representing 57.4 and 27.1% of the impacts,

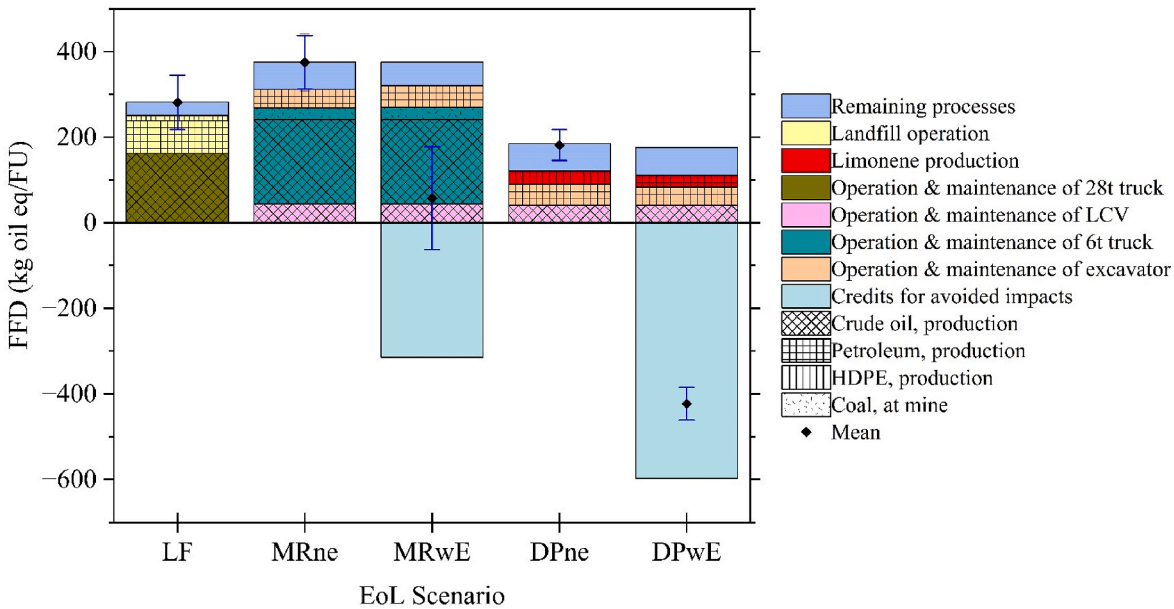

Fig. 7. Process contribution for fossil depletion potential at 1% cut-off. The black diamonds indicate the mean net impacts from Monte Carlo analysis, with a positive number representing an adverse environmental impact and a negative number representing environmental benefits. This result showed significant impact to the fossil fuel reserve due to use of vehicles. Considerable savings from recovered PS were also observed in MRwE and DPwE.

respectively. For both MRne and MRwE, crude oil production associated with the operation of the 6t truck and LCV represented the majority of impacts, accounted for 52.2 and 11.9% of the total impacts. For MRwE, avoided impacts from PS recovery offered a negative contribution of −83.7% of impacts. For DPne, crude oil production during operation of LCV accounted for 22% of impacts, while petroleum production during excavator uses and D-limonene production represented 26.4 and 16.9% of total impacts. DPwE had similar process contributions, but credits for avoided PS production were −339% (Fig. 7).

4. Discussion

4.1. Life cycle impacts of EPS in pontoons

4.1.1. Non-EoL impacts

The results described in this paper presented the environmental footprint of EPS used in pontoons. Both the inventory analysis and impact assessment had clearly shown that the production stages, which includes raw material extraction and manufacturing activities, dominated most potential environmental impacts. This finding was not unexpected, as these stages are known to require an enormous input of non-renewable energy, such as crude oil and its derivatives. In addition to intense energy usage, the analysis showed that the production of virgin EPS also had adverse implications to air quality due to extensive use of pentane as a blowing agent during moulding.

The impacts of transporting personnel and equipment to the beach site were reasonably significant in this study. Impacts in this stage primarily stemmed from direct and indirect CO_2, SO_2, P, and N emissions due to fuel consumption and air emissions. The transportation of heavy machinery to/from the site constituted the majority of impacts during this stage. This corresponded well with the initial assumptions regarding the loads and travelling distance. Additionally, the dataset used when modelling this stage was modified to fit the transport requirements for this case study. Thus, the life cycle impacts of transportation in this study were higher compared to models that assumed the default containerised transportation. When considering the use of heavy machinery, the results showed that this stage contributed 4–12% of all impact category, which is relatively small compared to other life cycle stages. Most of these impacts were due to direct emissions from fuel combustion, with the excavator being the primary contributor.

4.1.2. EoL impacts

The discussion on midpoint LCA results from here on emphasises EPS EoL disposal/recycling, as the preceding stages were identical for all scenarios. Analysis of the LF scenario showed that impacts in GWP, TAP, and FFD were primarily attributed to the use of the 28t truck. Particularly, fuel combustion in this vehicle to transport the pontoon to landfill substantially increased CO_2 and SO_2 emissions. However, this impact is still considered to be low compared to another study by PWC (2011). Particularly, this study modelled the transport of empty EPS boxes and reported impacts in GHG emission, SO_2 emission, and oil consumption to be 4.7, 4.2, and 3.6 times higher than our results, respectively. The main reason for this disparity can be due to differences in initial assumptions, particularly regarding payload and transport distance. Regardless, findings from PWC (2011) and this study showed that impacts from transporting EPS cannot be ignored and strategies for on-site size reduction must be adopted not only for pontoons but all EPS wastes.

It should be noted that even without this transport impact, the environmental implications of operating the landfill were still considerably higher than other LCA publications with similar premises in climate change, acidification, and eutrophication potentials. Particularly, our results were at least one degree of magnitude greater than results from other studies (Table 4). This can be explained by the fact that previous studies allocated landfill infrastructure burden based on mass, which may work well for waste streams with a density close to 1000 kg/m^3, but not for bulky materials such as EPS. Thus, previous studies may underestimate the potential impacts of landfilling EPS materials. To address this, we calculated impacts based on volume, which allowed us to accurately take into account the amount of landfill space occupied by EPS wastes.

The use of on-site MR provided a pathway to reduce the volume of EPS waste, and therefore, should have minimised the impacts of transportation. However, as seen in Figs. 3–5 and 7, this was not the case, as the use of the 6t truck to transport the densifier, which weighs up to 3.2 tonne, to/from the site significantly added to the overall impacts of both MRne and MRwE scenarios. Additionally, the use of LCV to transport the densified PS ingots to a landfill/recycler further increased the impact of MR scenarios. Thus, impacts stemming from transportation during MR

Table 4
Landfill impact scores of this work compared to selected studies from the literature. Due to differences in functional units, the impacts scores quoted were standardised to "per kg waste" to enhance comparison.

Publication	Plastic type	Method	Impact category		
			Climate change (kg CO_2 eq)	Acidification (kg SO_2 eq)	Eutrophication (kg N eq)
This study (incl. 28t truck)	EPS	ReCiPe Midpoint (H)	1.2×10^{0}	5.8×10^{-3}	6.3×10^{-3}
This study (excl. 28t truck)			2.8×10^{-1}	1.3×10^{-3}	6.1×10^{-3}
Lindstrom and Hicks (2022)		TRACI	1.4×10^{-1}	2.2×10^{-4}	1.6×10^{-2}
Lim et al. (2021)		TRACI	6.6×10^{-2}	1.9×10^{-4}	4.0×10^{-3}
Hossain et al. (2021)	Mixed plastics	IMPACT 2002+	7.4×10^{-2}	N/A	N/A
Gear et al. (2018)		CML	1.0×10^{-1}	2.8×10^{-4}	N/A
Xayachak et al. (2023)		ReCiPe Midpoint (H)	1.1×10^{-1}	7.4×10^{-5}	1.3×10^{-3}

scenarios can be considerably reduced by using lighter, more mobile thermal densifier models. For GWP, the benefits from recovered PS were insufficient to offset these EoL impacts, even when considering data uncertainty. Similarly, analysis showed that the environmental burden associated with MR was not balanced by recovered PS in MEP and FFD categories. However, Monte Carlo analysis results showed that it is possible to achieve net savings in MEP and FFD using MRwE scenario. This would require taking measures to increase the substitution factor of the recovered PS ingots (e.g., minimise contaminants that may decrease the purity of recovered PS products). Significant benefits were observed for MRwE in TAP due to avoided SO_2 emissions associated with fuel use during PS production. Based on the results of this study, MR of EPS using thermal densifiers seemed to only provide better environmental performance than the LF scenario if PS ingots were recycled.

LCA results for DP indicated that the energy and material input for DPne and DPwE imposed the least impact in most categories (including data uncertainty) except for FEP. High P emissions that contributed to FEP stemmed from the assumptions that wasted D-limonene was treated as a hazardous waste due to its flammability. This impact could be minimised by increasing the number of cycles that D-limonene was used to dissolve EPS, such as taking measures to decrease oxidation during storage and transportation. The high uncertainty observed in Fig. 5 reflected the potential benefits if this solvent was used for more cycles. Regardless, this result highlighted the need to consider appropriate limonene disposal options if this approach was selected to treat EPS waste in pontoons. Another prominent contributor to P emission was due to the combustion of heavy fuel during sea transport. Thus, there are opportunities for this impact to be reduced if Australia-made D-limonene was used.

The avoided impact from recovered PS made DP even more environmentally favourable. As a result of the selective nature of limonene and the lack of heat input, minimal polymer degradation would take place (Achilias et al., 2009). Thus, recovered PS in the DPwE scenario offered greater environmental savings than MRwE. This was shown in GWP, TAP, MEP, and FFD, where credits for avoided impacts outweighed the impacts from energy and material input. It should be noted that this benefit was dependent on the existence of re-processor capable of (1) separating and recovering PS and limonene; and (ii) recycling the PS into new products. However, considering the degree of contamination of EPS in pontoons, the lack of supporting infrastructure, the absence of local manufacturing industries, and inadequacy in regulatory framework, recovery of PS from pontoons can be extremely difficult. Overall, the result, which showed DP approach having the best environmental profile, was unexpected as DPne and DPwE were given the worst-case data found in the literature. This implied that in a real-world settings, where optimal amounts of D-limonene were used and leakage/waste was minimised, the impacts from this stage could be even lower.

4.2. Sensitivity analysis & limitations

Despite best efforts, data uncertainty is inevitable in LCA. The errors bars in Figs. 3–7 represent the results from Monte Carlo analysis. The uncertainty indicated a relatively wide range of impact scores for MRwE in GWP, DPne and DpwE in FEP, LF in MEP, and MrwE in FFD. The primary causes for this were due to uncertainty in substitution factor (for MrwE), quantity and disposal method of D-limonene waste (for DPne and DPwE), and landfill infrastructure burden (for LF). Despite these uncertainties, the conclusion derived from this study, which favoured using on-site DP for size reduction prior to transportation, was relatively stable.

This study had several modelling limitations that should be considered for future works. First, this investigation assumed that EPS used in pontoons is equivalent to general-purpose EPS, which may not be entirely accurate. Second, our model did not account for the potential release of EPS due to logistical delays, the duration of which can differ amongst EoL scenarios (e.g., LF may have longer delays due to the need of large hauling trucks, leading to more EPS being released into the sea). Additionally, EPS leakage during transportation to landfill and removal from pontoon was not taken into consideration. These exclusions may lead to underestimating the environmental impact of managing missing pontoons. Third, the operation of the thermal densifier was based on limited data from the manufacturer, which might be biased and not representative of the actual environmental impact of the process. Similarly, data for D-limonene production was based on experimental work, which might not accurately reflect real-world manufacturing. Additionally, the quantity of D-limonene used was based on studies that used EPS from packaging industry, which may have different properties compared to EPS used in manufacturing aquatic products. Furthermore, the current study could not model the impacts associated with the cautionary measures needed to manage the inherent flammability of D-limonene during transportation and disposal. Fourth, this work did not account for Australia's changing energy profile, which is gradually transitioning into more sustainable fuel sources and vehicles, such as electric and hydrogen cars and trucks (Albatayneh et al., 2023). Lastly, the authors recognised that recovery scenarios modelled in this study (MRwE and DPwE) represent idealistic settings for which we should strive. However, this is not realistically achievable with current technologies and political climate.

4.3. Recommendations

The results from the LCA analysis clearly favoured the use of on-site dissolution/precipitation method to increase the efficiency and minimise the environmental impacts of managing EPS waste from flood-released pontoons. However, it is important to acknowledge that this option is highly dependent on the availability of a recycling/chemical plant with equipment capable of processing D-limonene-PS mixture. Based on our current knowledge, such facilities have yet to be developed in Queensland. Landfill disposal was secondarily preferable in terms of life cycle impact, followed by on-site mechanical reprocessing. Currently, landfill disposal of pontoons seemed like the most practical approach as it does not require substantial changes to the current waste management system. However, there is limited social consensus to open new landfills due to its inherent lack of circularity. Consumers are gradually becoming more aware of the ecological damages of EPS not only in aquatic structures but also in everyday products (O'Farrell et al., 2021). Thus, key players in the plastic industries and waste management, such as manufacturers, re-processors, governmental entities, landfill operators, and NGOs must be prepared for increasing demands for environmentally-responsible waste management and manufacturing practices.

Presently, the recovery of PS from flood-released pontoons is not feasible due to the risk of contaminants entering and damaging the recycling equipment. Contaminants may also reduce the quality of recovered material, limiting its applications. Additionally, Queensland (and Australia) has yet to develop a sustainable market for recycled PS pellets (O'Farrell et al., 2021). Thus, local manufacturers have limited financial incentives to use inferior recycled plastics. Recovery efforts are further impeded by the lack of investment in research, low commercialisation of recycling programs, and inadequacy of infrastructure for collecting, sorting, decontaminating, and recycling of EPS wastes (Hossain et al., 2022). Addressing these issues through an appropriate combination of stringent regulations, policy and structural reforms, investment in infrastructure, and incentive programs can significantly minimise the impacts of plastic waste management and provide an avenue for EPS waste to be recycled.

Considering the current limitations that prohibited the recovery of PS from pontoons and other aquatic products, future efforts in managing marine litter can significantly benefit from mandated recycling-oriented designs. In other words, the recoverability of materials needs to be considered during the design phase. This can include separating the plastic components from the metal and concrete structure of the pontoon to minimise contamination. Additionally, the use of recycled materials should be incentivised.

Increased investments in research and development for emerging plastic recycling technologies can help diversify our waste management infrastructure, providing more avenues by which EPS wastes can be efficiently recycled and managed. Notably, the selection of D-limonene did not imply its superiority over other solvents, but rather to demonstrate the potential environmental benefits of using natural-based solvent for EPS waste management. Thus, there is opportunity to expand this work to investigate optimal working parameters that minimises impacts from applying DP technologies for EPS waste management. This includes identifying the most appropriate/efficient natural solvent (D-limonene, myrcene, and *p*-cymene, etc.), minimum solvent consumption rate, optimal separation processes, and the most environmentally favourable solvent disposal method. Given that DP scenarios had the lowest impacts despite using worst-case assumptions and data, it is reasonable to assume that impacts from an optimised DP system can be even lower. Additionally, there is a scope for "what-if" LCA modelling with particular consideration given to Australia's transition to more renewable energy and fuel sources. Results from this type of analysis can be highly valuable in promoting sustainable waste management strategies. Lastly, this paper proposes the development of techno-economic analysis (TEA) in conjunction with LCA modelling to help identify the most environmentally and economically feasible method of managing EPS wastes.

It is worth noting that resolving the issue of EPS pollution by simply focusing on EoL processing is not a sustainable approach. As with other waste management strategies, the waste hierarchy must be followed. That is, the generation of waste must be minimised/avoided first and foremost. For example, materials that are less likely to fragment, such as expanded polypropylene (EPP) (Colombie, 2017) and air-filled HDPE (Fauna and Flora International, 2020), can be used to replace EPS in future production of pontoons and buoys. However, these materials are still plastics, and their uses must be accompanied by comprehensive life cycle assessments that include the ecological impacts of microplastic to the marine environment. Additionally, promoting appropriate care of pontoons can minimise the impacts of EPS on the environment. This includes providing guidelines for better maintenance, securing or removing EPS-containing products during turbulent seas.

5. Conclusions

This work investigates the environmental profile of different management scenarios of expanded polystyrene (EPS) in flood-released pontoons due to a recent flooding event in Queensland. Applying the LCA framework, this study found that the production stages of EPS, which includes raw material extraction, refinement, and manufacturing activities, had the most significant environmental footprint. Impacts due to landfill disposal of EPS were considerably high due to the bulky nature of EPS, leachate emission, and the use of large hauling trucks to transport full-size EPS to the landfill.

The impacts from landfill scenario highlight the need for on-site size reduction of ES prior to transportation to the waste management facility. Mechanical re-processing using thermal densifier and dissolution/precipitation using D-limonene are two possible solutions for this. The analysis found that mechanical re-processing did not provide any environmental benefits as the burden of transporting the thermal densifier to/from outweighed the potential savings. Conversely, the use of D-limonene to dissolve EPS had the least impacts, indicating future opportunities for dissolution/precipitation method to be used for treating EPS wastes. PS recovery from pontoons are theoretically possible and can provide substantial environmental benefits. However, this is not presently feasible due to risks of contaminants entering and damaging the recycling machineries, lack of sustainable market for recycled materials, and inadequate infrastructures.

This study also identified and addressed a several limitations from previous publications. Particularly, care must be taken when modelling voluminous materials as they tend to occupy more spaces in landfills and transport vehicles than municipal solid wastes. Additionally, previous LCA studies centred around the management of EPS waste from the packaging industry with a fixed collection point and pre-defined logistics. Thus, the novelty of our work lies in the development of LCA models that specifically accounted for (i) the low-density properties of EPS; and (ii) variation in transportation distances to/from collection sites.

This study also identified several limitations upon which future works can improve. This includes using real-world, measured data and accounting for EPS leakage during transport and handling. Additionally, it was recommended that end-of-life disposal should not take priority over waste avoidance. The latter can be achieved by using alternative materials for pontoon production and promoting improved care/responsibility of pontoons and other aquatic products, especially during turbulent sea.

Credit author statement

Tu Xayachak: Conceptualization, Data curation, Formal analysis, Investigation, Methodology, Validation, Writing - original draft. Nawshad Haque: Conceptualization, Supervision, Writing -review& editing. Deborah Lau: Conceptualization, Supervision, Writing -review& editing. Nargessadat Emami: Conceptualization, Writing -review& editing. Lincoln Hood: Conceptualization, Writing -review& editing. Heidi Tait: Conceptualization, Writing -review& editing. Alison Foley: Conceptualization, Writing -review& editing. Biplob Kumar Pramanik: Conceptualization, Project administration, Supervision, Writing - review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

No data was used for the research described in the article.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at https://doi.org/10.1016/j.chemosphere.2023.139400.

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