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LCIA OF IMPACTS ON HUMAN HEALTH AND ECOSYSTEMS

Expanding life cycle impact assessment to account for marine plastic emissions: a case study for the fishing industry

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Abstract

Purpose Fishing is a major source of plastic pollution, which is harmful to marine life. Life cycle assessment studies do not usually include plastic emissions in their inventory or impact assessment. There has been a recent surge of methodological development to include plastic impacts in life cycle impact assessment (LCIA). This study of anchoveta and cod fishing included dissipative plastic emissions in the inventory data and tests currently available characterisation factors for plastic pollution impacts to identify their importance for ecosystem quality damage.

Methods Life cycle inventory (LCI) data was gathered for two systems that produce 1 tonne of fish product (anchoveta fishmeal, consumer packaged cod). LCIA calculations were performed using IMPACT World+ to calculate ecosystem quality damage. In addition, plastic impact pathways for the physical effects from microplastics on biota (both water and sediment-dwelling organisms) and entanglement were included.

Results and discussion The fishing stage dominates the ecosystem quality (EQ) damage results for both systems. For cod fished with gillnets, production of antifouling coating is the dominant contributor. Entanglement results completely dominate the picture for anchoveta. Overall, freshwater ecotoxicity (long term) causes the largest overall EQ damage. The emissions that contribute the most to freshwater ecotoxicity (long term) are metals from fishing gear production (mainly associated with the lead in the bottom rope), followed by polyethylene production for the cod system. For anchoveta, production of phosphoric acid and iron sulphate for fishmeal processing contribute the most. The second most important impact category is climate change (long term), due to emissions of fossil-derived carbon dioxide, as well as refrigerant R500 for anchoveta. Even with 100% fragmentation into microplastics, physical effects on biota account for less than 0.1% of the total EQ damage.

Conclusions Including the impacts associated with plastic emissions is important in LCAs where plastic materials or products are used. Without the inclusion of the plastic impacts, LCAs will not capture all of the impacts that plastics can have on the ecosystem. This study shows that whether parts, or the whole of the gear (including ropes) are included affects the importance of the mismanaged waste losses in the value chain, as well as the microplastic impacts associated with fishing gear losses at sea. The timescale for when lost fishing gear can still entangle organisms is extremely important for the overall damage caused by fishing and should be investigated further.

Keywords Marine plastic · Fishing · Life cycle impact assessment · Anchoveta · cod · Microplastics

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1 Introduction

Fishing is a major source of plastic pollution (Deshpande et al. 2020) and industrialised fishing nations' aquaculture and fishing industries are responsible for large amounts of plastic in the oceans (Lebreton et al. 2022). Plastic has been found to be harmful to marine wildlife and damages marine ecosystems (Coyle et al. 2020; Thushari et al. 2020; Chen et al. 2021). However, environmental management methodologies, such as Life Cycle Assessment (LCA), have repeatedly ignored the leakage of plastic fractions to the biosphere in their inventory modelling (Boulay et al. 2021). Data and developments made within the PlasticLeak project (PLP, now carried on through the Plastic Footprint Network, PFN) are addressing the Life Cycle Inventory (LCI) challenge. In parallel, the MarILCA project, supported by the United Nations Life Cycle Initiative, addresses Marine Impacts in LCA and has been developing knowledge and methods to include plastic litter in LCA (Woods et al. 2021; Boulay et al. 2021).

Life Cycle Impact Assessment (LCIA) for plastics is described in Woods et al. (2021) and Askham et al. (2023). Plastic is emitted as either macro-, micro-, or nanoplastic, as reported in the LCI. It undergoes fragmentation and degradation as it passes through or accumulates in different environmental compartments, as characterized by the fate factor (FF). The ecosystem exposure factor (XF) characterizes exposure occurring through several exposure routes and pathways. The effect factor (EF) combines exposure-response and severity to yield damages to ecosystem quality. Characterisation factors (CFs) are the product of the FF, XF and EF, that are used to calculate the environmental impact of an LCI flow. Developments in LCIA have led to new CFs to either improve existing impact categories or to construct new impact categories based on new impact pathways (Wu and Su 2020). Despite these efforts, most of the currently used assessment methods include only very few impact pathways related to marine resources and ocean conservation (Ruiz-Salmón et al. 2021).

The lack of impact pathways representing environmental damage linked to marine habitats has led to a recent new surge of methodological development which aims to assess marine environmental impacts, such as biotic resource depletion (Hélias et al. 2023; Stanford-Clark et al. 2024), seabed impact (Woods and Verones 2019), effects of marine invasive species (Gjedde et al. 2024) and a marine plastic pollution framework (Woods et al. 2021). Models with different levels of completeness (from full CFs to individual fate and effect factors) have been published for plastic pollution. These include a range of sizes (plastic) and impacts, such as for physical effects on biota of microplastics ($>1\,\mu m \leq 5\,mm$; Lavoie et al. 2022; Corella-Puertas et al. 2023; Hajjar et al. 2025; Saadi et al. 2025) and macroplastics ($>5\,mm$)—through entanglement (Høiberg et al. 2024), as well as for ecotoxicity of plastic additives (Casagrande et al. 2024; Tang et al. 2022). However, CFs for macro- and microplastics have not yet been combined in an LCA, so both the relative impacts of micro- versus macroplastics and their contribution compared with traditional LCIA impact categories are unknown. Physical impacts on biota from microplastics are currently available in the released version of the Global Guidance for Life Cycle Impact Assessment Indicators and Methods (GLAM) LCIA method (Saadi et al. 2025; Life Cycle Initiative 2025).

The proliferation of these plastic CFs and ultimately impact categories is necessary to close the gap in terms of representativity for marine ecosystems. In parallel, though, the existence of an increasing literature of LCI elementary flows that can represent the release of plastic emissions, either in the form of macro- or microplastics, to the ocean, is needed for these CFs to be operational in life cycle analysis (Maga et al. 2021). The PFN has been the main initiative in the field, publishing a set of calculation frameworks to estimate plastic release for different sectors (e.g., transport, textile, or packaging) (Plastic Footprint Network 2024), providing information on the leaked plastic mass and redistribution into different environmental compartments. In contrast, only a small set of scientific articles have delved into plastic releases from specific sectors, by providing emission factors (e.g., Salieri et al. 2021; Loubet et al. 2022; Deville et al. 2023). The fishing industry has been shown to be an important source of plastic emissions worldwide (Schwarz et al. 2019). These emissions occur directly to the ocean through (intentional or accidental) loss of fishing gear (Kuczenski et al. 2022), wearing of antifouling coatings (Kim et al. 2024), as well as wear and tear and weathering losses from the fishing gear and aquaculture infrastructure during use (Grimaldo et al. 2023), among others. Other emissions occur on land, such as during terrestrial processing stages, or from tyre wear, as well as other coastal (Loubet et al. 2022) or terrestrial activities that generate plastic emissions that will ultimately reach the ocean (Sanchez-Matos et al. 2024; Marhoon et al. 2024).

Therefore, the main objective of this study was to test the currently available CFs dealing with plastic pollution impact categories in LCIA by conducting two case studies in the fishing industry, one providing an industrial fishmeal product and another providing a household consumer food product, accounting for direct and indirect human consumption systems. The two products involve a different type and intensity of plastic packaging (per unit product), as well as different types of fisheries in different geographic locations. The aim of studying these two different, but related systems is to include dissipative plastic emissions and identify their importance in terms of damage to ecosystem quality in comparison to traditional LCIA impact categories, that often do not include effects on the marine ecosystem (e.g. IMPACT World+ v1.05 and ReCiPe 2016). Therefore, this paper provides the first case studies that consider impacts of both micro- and macroplastics leakage in LCA. Further, while previous studies have quantified emissions of plastics into the environment from fisheries (Loubet et al. 2022), they were not combined with the recently developed CFs to assess the impact of leaked plastics. Thus, these cases will illustrate whether inclusion of these plastic impacts would suggest any changes in the decision-making process when interpreting LCA results and will inform policy making with regards to plastic waste management by identifying hotspots for marine plastic impacts in the context of seafood production. The two case studies are the landing and processing of artic cod (*Gadus morhua*) for human consumption in Norway and the landing of Peruvian anchoveta (*Engraulis ringens*), and its processing into fishmeal for export to Norway as an important ingredient in feed for aquaculture.

2 Methods

2.1 Goal and scope

The function of the production systems was to deliver a given amount of seafood product to Norwegian consumers. The two cases are different, as one supplies fishmeal as feed to the aquaculture industry and the other a cod food product to household consumers. The functional units for the two systems are:

- *Distribution of 1 tonne of packaged anchoveta fishmeal, supplied to a Norwegian fish farm, including waste management of user packaging.*
- *Distribution of 1 tonne of packaged cod supplied to Norwegian household consumers, including waste management of user packaging.*

The reference flows for the two systems were 1 tonne of final product supplied to industry in 50 kg polypropylene bags for the fishmeal case, and 1 tonne of cod supplied in 400 g units to household consumers. The cod packaging consisted of plastic bags made from nylon and low-density polyethylene, contained in cardboard boxes.

The system boundaries (see Fig. 1) for both cases include fishing, processing and packaging of the product, as well as the end-of-life for the packaging. This study includes the usual LCI accounting for material and energy flows, as well as emissions to soil, water and air, but goes beyond this to include aquatic macro- and microplastic emissions throughout the supply chains modelled.

A brief description of the two systems follows, with more detail given in the Life Cycle Inventory section (2.2). The anchoveta fishery uses purse seine equipment, catching anchoveta in Peruvian fishing grounds and returning the catch to port in Peru. The fishmeal is also produced in Peru, before shipping to Norway in large plastic sacks. The fishmeal is used as one ingredient in the feed for fish farmed

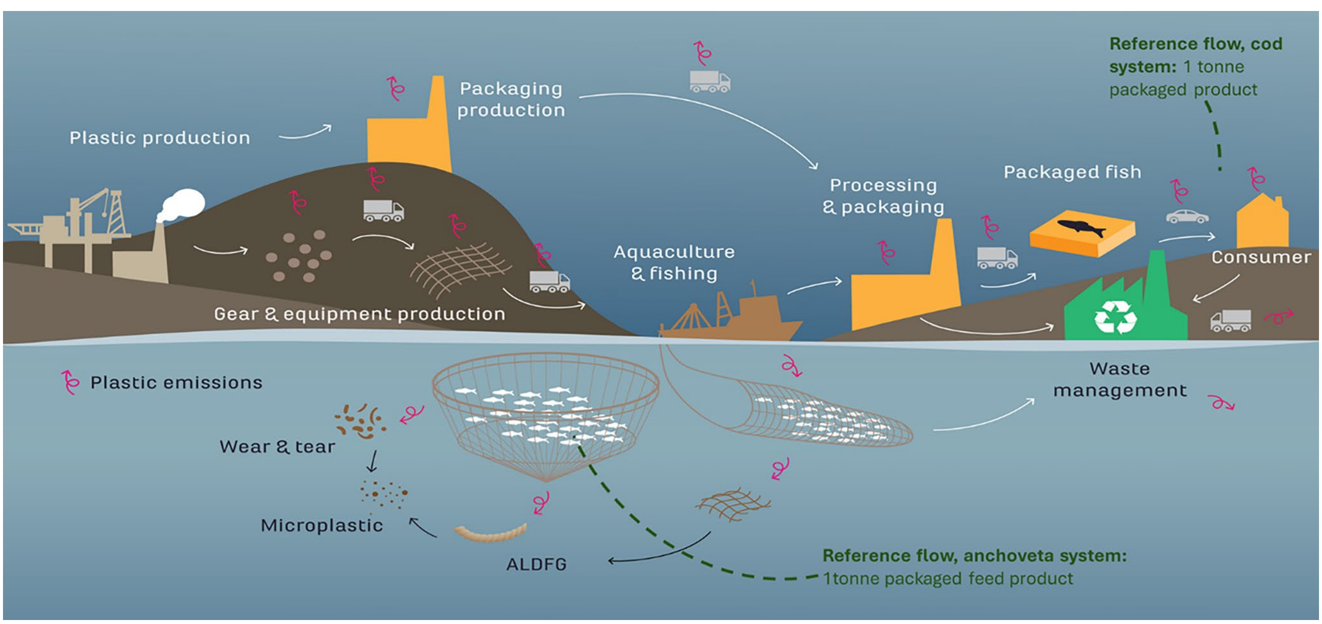

Fig. 1 System illustration including plastic emissions for two fishing industry cases, cod fishing (for human consumption) and fishmeal (feed for fish farms) for the Norwegian market. The reference flows for the two systems are shown in green text. ALDFG stands for abandoned, lost or discarded fishing gear

in Norway. Cod is caught in gillnets in Norwegian waters, returned to port and processed and packed before transport to shops for sale to household consumers. The consumers drive home with their shopping and consume the fish, discarding the packaging in their municipal waste management system for food packaging. The allocation perspective used for allocating emissions between fish products (e.g. fishmeal and fish oil) is mass allocation, although economic allocation is also explored.

2.2 Life cycle inventory - LCI

The data and references used for the different life cycle stages are shown in Table 1 and described in the following subsections. Where ecoinvent data is used (see Table 1), version 3.8 was used for the anchoveta case and version 3.10 was used for the cod case.

2.2.1 The anchoveta system

In the fishing stage, microplastic emissions were estimated for the use of antifouling, and macro- and microplastic emissions were estimated from the use of fishing gear (Sundt et al. 2018; Syversen et al. 2020). Plastic pollution emitted to the ocean due to hull antifouling paint is estimated to be 6% of the total weight of paint applied (1% due to weathering and 5% during maintenance) annually (OECD 2009). Plastic emissions during anchoveta fishing were estimated using the emission factors from Deville et al. (2023), which were 373.1 g/t catch, based on a 7% emission factor for purse seiners from Richardson et al. (2019). When the purse seine fishing gear from vessels belonging to Peruvian fishmeal and fish oil (FMFO) companies reaches its end-of-life, the main component of these gear (i.e., nylon) is sent to recycling to be used in toy and clothing manufacturing industries. In contrast, third-party vessels are expected to dispose of their fishing equipment in landfills or open dumps, and most sites along the central and northern Peruvian coast are expected to be open dumps (Ziegler-Rodriguez et al. 2019). Factors for mismanaged waste leading to emissions of plastics to the ocean from gear waste management are from PFN (Plastic Footprint Network 2023a, the emission factor for mismanaged waste to the ocean is 7.6%).

Plastic emissions also arise from mismanagement of the packaging that the anchoveta fishmeal is in when transported to the port of Callao in Peru, prior to shipping to Norway. These emissions (from packaging mismanagement) are estimated using PFN (Plastic Footprint Network 2023a, the emission factor for mismanaged waste to the ocean is 7.6%). 98% of all the Peruvian fishmeal in this study is transported by sea in bulk carriers, for which the plastic packaging used for land transportation is opened and the fishmeal is loaded into the cargo hold (Deville et al. 2025). This plastic packaging is disposed of in a landfill located in the metropolitan area of Lima and Callao. Plastic emissions arising from the disposal of packaging in the aquaculture stage are from the disposal of the packaging that the anchoveta meal was supplied in. Mismanagement of this packaging waste in Norway (2.4%, of which 5% emitted to the marine compartment, see Table 2) and tyre wear particles from vehicle transport were included, based on the PFN guidelines (Plastic Footprint Network 2023a, b) and Quantis and EA (2020). Emission factors for tyre wear in Peru, based on the references listed, are shown in Table 2. PFN (Plastic Footprint Network 2023b) provides the loss of tread as emission factor multiplied by share of tyre that is polymer, multiplied by the release rate to ocean, then this is divided by the average load. For further detail (including the emission factors used) see Table 2.

Allocation has proven to be a challenging methodological decision to consider in multifunctional systems in LCA (Pelletier and Tyedmers 2011). FMFO are two products that are jointly produced, which adds to the problem of partitioning the energy and mass flows in the production system. Therefore, as part of the discussion in this study, alternative allocation perspectives (i.e., economic and energy allocation) are included to produce fishmeal to understand how these choices can affect the final results presented for this particular case study. For mass allocation, the weights of both fishmeal (FM) and fish oil (FO, including both direct and indirect human consumption oils) were considered. Similarly, for the economic allocation, the average prices and production of both FM and FO in a three-year time frame (i.e., 2019 – 2022) (Banco Central de Reserva 2023a, 2023b) were considered to allocate the burdens. In contrast, for the energy allocation, information regarding the energy content was obtained from Fréon et al. (2017) with a proportion of 34:66 for FM:FO. Table 3 presents the different allocation of burdens for both products and all three methods.

2.2.2 The cod system

Emissions from onboard activities were not included for the Norwegian vessels, because any plastics used onboard were assumed to be collected and the waste managed onshore. The onshore management of this onboard waste is not included in the Norwegian case study. Antifouling emissions are based on Deville et al. (2023), as well as fishers' practices in Norway (i.e. antifouling coating is maintained each year, see Table 2 for the loss rates assumed for antifouling coating maintenance). Wear and tear gear losses (microplastics that are emitted directly to the ocean) were retrieved from Syversen et al. (2020). These losses were calculated per unit gear, based on the proportion of cod caught with gillnets and

Table 1 Summary of the steps included within the system boundaries for the two case studies

Elements included in the system boundaries of the analyses	Anchoveta case study	Cod case study
Fishing activity in which waters	Peruvian	Norwegian/Arctic
Type of fishery	Purse seine	Gillnet
Fishing gear cradle to fishing boat (resource extraction and gear production steps)[1]	Gear composition as in SI. Average nylon purse seine sizes for the industrial anchoveta fishery in Peru were considered. Lead weights were also included. Pellet losses during plastic production (Quantis and EA 2020 -PLP- and PFN) Losses from production, mismanaged waste rates from PLP & PFN.	Gear composition as in SI. All components listed in the SI were included, as well as their production and processing (metal working, extrusion, weaving, drawing and winding etc.). For all plastic components, pellet losses during plastic production are from PLP. Losses from production were included, with mismanaged waste rates from PLP & Plastic Footprint Network (2023b).
Vehicle transport in the value chain	Wear and tear tyres, Plastic footprint network (2023b).	
Fishing boat	Steel vessels. Primary data	Based on Winther et al. (2020). A demersal trawler is used as a proxy for all fisheries with a light weight of 3500 ton and an assumed lifespan of 30 years. Note, fishing vessel maintenance was not considered in Winther et al. (2020) also excluded here.
Antifouling	Based on Deville et al. (2023). Antifouling re-application/maintenance assumed 2.5 yrs in Peru.	Based on Deville et al. (2023). Production of the antifouling coating is taken from EPD, Jotun (Otterbech 2024). Fishers practices in Norway are such that the antifouling is replaced each year (personal communication, Jørgen Vollestad 6/6/2024).
Wear and tear of fishing gear during use	Richardson et al. (2019)	Syversen et al. (2020)
On board plastic emissions related to crew activities	Included (Deville et al. 2023)	Not included, assumed disposal at port.
Losses of fishing gear	Richardson et al. (2019)	Included, according to reported losses of fishing gear (Sundt et al. 2018). Unreported losses are not included.
Fuel use for fishing	Marine Diesel, on average around 29.1 l/t landed catch (primary data)	Included, coastal vessel data from Winther et al. (2020) for cod or similar species (e.g. haddock) using "conventional gear, such as gillnets" use 0.13 l/kg.
Refrigeration	R134a refrigerant. 2 kg per vessels, refilled, every 2.5 years. Third-party vessels considered without refrigerant	In Norway the gillnet boats fill their holds with ice in port. Thus, assumed as for ecoinvent process "Operation, reefer, freezing on land {NO}" which considers a reefer with a lifetime of 10 years and includes fuel and power consumption, maintenance, use of refrigerant (90% R134a and 10% carbon dioxide, liquid) and respective emissions. 0.2 kg ice/kg fish (primary data)
Waste disposal of fishing gear	Vessels from FMFO producing companies disposed of most of their fishing gear (i.e. nylon) to toy and clothing manufacturers. Third-party vessels are expected to dispose their fishing equipment in either landfills or open dumps.	Headrope and bridle lines (comprised of PP/HDPE) are assumed to be sent to recycling. 80% of collected PP/HDPE components end up being recycled and the remaining 20% are sent to landfill in Denmark. Footrope is assumed to be sent to landfill in Norway. Net is assumed to be sent to landfill in Poland. (Fenella Metz, personal communication, February 26, 2024)
Processing of the fish product	Anchoveta is processed in 4 FMFO plants located on the coast of Peru.	Winther et al. (2020) Light fuel oil and electricity, as well as detergent and infrastructure included.
Packaging of the fish product	Industrial consumer packaging included	Household consumer packaging included, i.e., a plastic bag in a carton box (Svanes et al. 2011)
Transport to the consumer	Road transport to main Peruvian port and maritime transport to importing country port. Road transport done with different types of motors (from unregulated to EURO 5). Maritime transport in wet and dry bulk carriers and container ship.	450 km from processing facility to retailer shop, road transport (Svanes E., personal communication, March 11, 2024); 0.54 km from retailer shop to household, road transport (Baxter 2022). To model the transport to the retailer (shop) the following ecoinvent process is used: Transport, freight, lorry > 32 metric ton, EURO6 {RER}| transport, freight, lorry > 32 metric ton, EURO6 | Cut-off. To model the transport between retailer shop and household the following ecoinvent process is used: Transport, passenger car, medium size, diesel, EURO 5 | Cut-off; Euro class and fuel type were selected based on Norwegian statistics (SSB 2023)

Table 1 (continued)

Elements included in the system boundaries of the analyses	Anchoveta case study	Cod case study
Waste management of the packaging	Included.	Included. Norwegian market processes from ecoinvent database were used to model packaging waste management: Market for waste polyethylene and market for waste plastic, mixture.
Mismanaged packaging waste	Included for activities in Peru, based on the PFN (Plastic Footprint Network 2024)	Included, based on the Plastic Litter Project factors appropriate for Norway (Askham et al. 2021) 2.4% of waste in Norway is mismanaged and 5% of this mismanaged waste will end in the ocean.

[1]The material composition of the fishing gear can be found in supplementary information.

the proportion of gillnet boats that are 11–14.99 m (Directorate of Fisheries, 2024), which gave 0.98 g/kg gillnet·yr. Norwegian fishers are encouraged to report fishing gear that is lost to the Norwegian Directorate of Fisheries (Vodopia et al. 2024); consequently, reported losses of fishing gear were included (Sundt et al. 2018), but unreported losses were not included. Using catch and vessel size data as for the wear and tear losses, the amounts of gear reported lost amounted to 1.38 g/kg gillnet. Gear waste management actors in Norway and gear recycling companies in Norway and Denmark were interviewed by the authors as part of the data gathering for the cod case. When the gillnets gear system has reached the end of life the waste management system in Norway is such that some gear is sent directly to a recycling plant in Denmark, some is sent to landfill in Norway, and some goes to other countries where it is separated. Some of this separated waste is recycled in Denmark and the remaining fraction is landfilled. Mismanaged waste rates from PFN (Plastic Footprint Network 2023a, 2.69% of which 5% released to the ocean) were used for calculation of emissions of plastic to the ocean arising from mismanagement of this fishing gear waste. The terrestrial supply chain stages up to the plate of a Norwegian consumer were included. Plastic emissions from mismanagement of packaging waste in Norway (2.4%, of which 5% emitted to the marine compartment) and tyre wear particles from vehicle transport were included (380 mg/km, 0.5 kg microplastic/kg tread, 17% release rate to ocean) (Plastic Footprint Network 2023b).

2.2.3 Plastic emissions in both systems

Plastic losses throughout the value-chain are in many different forms, such as terrestrial emissions like pellet losses from production (0.01%, 70% of which is assumed emitted to the ocean, Quantis and EA 2020 and transport of the granulate for gear production, and gear production waste. Some of the emissions in the LCI occur directly to the marine environment, such as fibres from wear and tear of ropes and nets, large pieces of net and the associated other plastic parts (e.g. buoys and floats) that can be lost during the fishing stage. Packaging litter and mismanaged waste plastic flows are also mostly initially on land. These inventory flows of plastic emissions can be in the form of macro- or microplastic (<5 mm). It was assumed that the wear and tear of gear was in the form of microplastic fibres and that between 50%-100% of macroplastics fragment to microplastics within a time-period of 100 years, in order to include their effects in this study. This is an oversimplification and ignores the temporal aspects of fragmentation of plastic. This will be further discussed, but is a limitation of this study (Pfohl et al. 2022; Hajjar et al. 2024; Kaandorp et al. 2023).

The LCI data for plastic losses per functional unit is provided in Table 4. Some allocation occurs when including plastic emissions throughout the two systems. Tyre wear particles for each road transport activity in the two life cycles studies are allocated to the transport load as for other emissions from road transport in ecoinvent (based on assumptions about load on the given vehicle). Plastic losses associated with plastic production and gear production were entered into copies of the ecoinvent processes and thus their allocation to the products produced follows the same approach as ecoinvent uses for these processes (see Tables 1 and 2). Gear losses (including wear and tear) and antifouling particles are allocated wholly to the fishing stage of the product system, but then during the processing stage, impacts for the upstream system are allocated according to the different products produced by the system. The basis model for both systems uses mass allocation for this. A sensitivity analysis is performed for this allocation for the anchoveta system in 4.2, where the authors compare the results if economic or energy allocation is used instead of mass allocation. Packaging losses for the specific product delivered to the customer in Norway are allocated wholly to the product system.

The losses included in the anchoveta system were retrieved from Richardson et al. (2019), based on a global meta-analysis that examined gear losses reported across scientific and gray literature from 1975 to 2017. However, the cod system gear losses are based on the losses reported to the Norwegian Directorate of Fisheries via a service called

Table 2 Loss rates used for plastic losses used in the system models

Inventory data	Anchoveta case study amount	Cod case study amount
Plastic production (for both gear and packaging)	For all plastic components. 0,01% microplastic bead waste was assumed for plastic production (Table 9–1, Quantis and EA 2020 Using release rates for small size particles and assuming 30% remains in river sediments and 70% is emitted to the ocean (Table 6–4, Quantis and EA 2020).	
Fishing gear production	Processing losses are dependent on the specific gear components and relevant process used for production. The inventory for plastic waste management already included in the relevant ecoinvent process was used as the basis for calculating the amount of mismanaged waste arising from the process. Thus, the amount of raw material discarded as waste was combined with the mismanaged waste rate from the "macro- from plastic waste" sheet (Quantis and EA 2020) for the specific production location for the given component (e.g. China) was used in combination with release rates for the amount released to water that reaches the ocean. This production waste is not microplastic beads in their initial form, but has been assumed to be beads for this calculation. Waste from plastic processing for gear components can be microplastic beads, fibres, or lumps of plastic that are larger. For this study, we applied the release rate for low residual value, small size (<5 cm), 40%. The release rate would be lower if assumed medium (5-25 cm) or large (>25 cm) plastic pieces. Note the PFN guidance is more specific about the type of product and splits into packaging, textiles and tyres. This kind of production waste does not fall into one of these categories, thus the generic plastic waste factors from the older study were used.	
Wear and tear fishing gear	The estimate of wear and tear from purse seiners was obtained from Richardson et al. (2019), for which 6.6% of all the gear is expected to leak into the ocean.	Syversen et al. (2020) provide an annual estimate of the wear and tear plastic emissions (fibres) from gillnet fishing of 9.2 tonnes/year. This data has been combined with relevant annual catch data for cod caught in gillnets (Directorate of Fisheries, 2024) to provide the data for wear and tear per functional unit.
Antifouling	Plastic polymer content was assumed to be 25%. The amount of anti-fouling paint needed was calculated as described in Deville et al. (2023). The emissions of antifouling paint to the ocean were as follows: 1% due to weathering and 5% during maintenance (3.2% during abrasive blasting and 1.8% during painting).	
Packaging	Woven propylene bags are used for packaging with an average of 3.21 kg per ton of fishmeal. It is expected that 67% of textiles (macro) waste is mismanaged in Peru and approximately 7.6% is expected to reach the ocean (Plastic Footprint Network 2023a). Plastic production losses from granulate production, or packaging production processes have not been included for the consumer packaging or shrink wrap in this study.	Shrink film used around pallets assumed to be mismanaged as for other plastic packaging waste in Norway: 2.4% of waste in Norway is mismanaged and of this mismanaged waste 5% would end in the ocean (Quantis and 2020, n.b. this reference is used, as; Plastic Footprint Network 2023a does not include mismanaged waste factors for Norway). Plastic production losses from granulate production, or packaging production processes have not been included for the consumer packaging or shrink wrap in this study.
Tyre wear particles for lorry transport	Plastic Footprint Network (2023b) Loss of tread = (emission factor) ×share of polymer ×release rate to ocean)/average load Emission factor = 380 mg/km. Share of polymer = 0.5 kg microplastic/kg tread (transport of goods, heavy). Release rate, RR: 17% ocean due to the arid conditions of the Peruvian coast and the closeness of the roads to the ocean, all of the 15% that is expected to be released into freshwater is expected to reach the ocean.	Plastic Footprint Network (2023b) Loss of tread = (emission factor) ×share of polymer ×release rate to ocean)/average load Emission factor = 759 mg/km (central value, heavy). Share of polymer = 0.5 kg microplastic/kg tread (transport of goods, heavy). Release rate, RR: 2% ocean + 15% freshwater. However, only 10% of the 15% that goes to freshwater actually ends up in the ocean (column T, rows 115 to 120, Plastic Footprint Network 2023b). Therefore, RR(ocean) = 2 + 1.5% = 3.5% Average load of a > 32 ton lorry size class is 15,960 kg (provided in the documentation sheet)
Tyre wear, passenger car	Not relevant	Plastic Footprint Network (2023b) Loss of tread = emission factor ×share of polymer ×release rate to ocean)/average load. Emission factor = 80 mg/km (central value, passenger car). Share of polymer = 0.35 kg microplastic/kg tread (transport of passengers, passenger car). Release rate: 2% ocean + 15% freshwater. However, only 10% of the 15% that goes to freshwater actually ends up in the ocean. Therefore, RR(ocean) = 2 + 1.5% = 3.5% According to the documentation sheet, "The average weight of this category is estimated to be 1600" (medium size car).

Table 2 (continued)

Inventory data	Anchoveta case study amount	Cod case study amount
Mismanaged waste – fishing gear	Fishing gear of industrial anchoveta fishery weight on average 41 ton per vessel and is expected to be managed differently as to other waste. Fishing gear of industrial fishing companies in Peru are expected to be sold to companies for recycling into clothing and toys.	Footrope is landfilled in Norway. This is made of PP, HDPE and lead. The plastic part of the rope that is mismanaged is thus (footrope weight-losses due to wear and tear)*0.5. Mismanaged waste rate for macro textile waste in Norway is 2.69% (Plastic Footprint Network 2023a), 5% of large items with low residual value are released to the ocean (all countries). For the proportions of waste that are sent to recycling in Denmark and Norway the following amounts were considered to be landfilled: 20% plastic components sent to recycling in Denmark end up in landfill (in Denmark) and 7% plastic components sent to recycling in Norway end up in landfill (in Norway). The mismanaged waste rate for macro textile waste in Denmark is 0025%.
Mismanaged waste – packaging	Plastic packaging of fishmeal is landfilled in Peru and Norway. The fishmeal bags that have to be cut in order to load the fishmeal to cargo hold of bulk carriers are landfilled in Peru (approximately 54.4 tons). In contrast, the fishmeal that is transported via container ships keeps its packaging to then be landfilled in Norway (approximately 1 ton). It is expected that 67% of textiles (macro) waste is mismanaged in Peru and approximately 7.6% is expected to reach the ocean (Plastic Footprint Network 2023a).	2.4% of waste in Norway is mismanaged and of this mismanaged waste 5% would end in the ocean (Quantis and 2020, n.b. this reference is used, as; Plastic Footprint Network 2023a does not include mismanaged waste factors for Norway)

Table 3 Energy, economic and mass allocation for Peruvian fishmeal (FM) and fish oil (FO) production in 2021 (Deville et al. 2025)

Allocation	FM	FO direct human consumption	FO indirect human consumption
Energy	34.00%	58.22%	7.78%
Economic	83.04%	16.96%	
Mass	88.51%	10.29%	1.20%

FiskInfo (https://www.barentswatch.no/fiskinfo/). Fishers log gear in use and any lost gear on this platform, as well as lost gear they find. Using this service is obligatory under Norwegian law (Harvesting Regulations 2021: §69). This also applies to parts of equipment, such as trawl ropes. If it is not possible to have equipment picked up, it must be immediately reported to the Coast Guard with information on the type and amount of equipment, as well as the location where the equipment was last seen. Vessels with an electronic catch logbook can also report lost gear through the logbook. Richardson et al. (2022) describe possible reluctance to report gear losses in interviews due to overall negative attitudes and repercussions associated with reporting perceived high levels of gear losses. This means that it is possible that the Norwegian data for reported gear loss underestimates the actual losses in the cod system.

2.3 Life cycle impact assessment

IMPACT World+ was the selected LCIA method used for calculating ecosystem quality results in this study, at both midpoint and endpoint. The version implemented in the SimaPro 10.1 software in March 2025 was used for the calculations presented in this paper. In SimaPro this IMPACT World+ version is described as corresponding to IMPACT World+ Endpoint version 1.47, including only recommended (not interim) damage level indicators and partial regionalization. For the Ecosystem Quality damage, this LCIA method aims at quantifying a local and temporal loss of a fraction of species, measured in PDF.m^2.yr (Bulle et al. 2019).

IMPACT World+ was the chosen LCIA method because it provides a comprehensive coverage of ecosystem quality impacts, including both short- and long-term perspectives. The microplastics CFs were readily available for use in units compatible with IW+ at the time of conducting the study, while the GLAM LCIA method was not yet operational. Further, other case studies that included impacts of leaked plastics were conducted using IW+, therefore, using the same LCIA methods allowed for comparison.

In addition to these IMPACT World+ calculations, some novel emerging methods were used to include different impacts arising from plastic losses in the two value chains studied, as described in 2.3.1.

2.3.1 Plastic impact pathways

Impact pathways linked to the physical effects from microplastics on marine biota (both water- and sediment-dwelling organisms) and entanglement (a physical effect of macroplastic on biota) were included. CFs for microplastics from Saadi et al. (2025) consider the fate, exposure and effect of microplastics in both marine water and sediments. The fate was assessed by considering the transfer between each compartment (sedimentation and resuspension) and loss rates

Table 4 Microplastic emissions per functional unit

g microplastic/FU	Anchoveta case study amount		Cod case study amount	
Fragmentation rate	50%	100%	50%	100%
Fishing gear production				
EPS beads			7.5	7.5
HDPE beads			7.1	7.1
PA beads			45.5	45.5
LDPE film fragments	0.002	0.005		
PA microfibres	0.35	0.69		
PP beads			14	14
TRWP			0.009	0.009
Use incl. wear and tear and antifouling				
EPS beads			79.2	111.7
HDPE microfibres			103.8	157.7
HDPE beads			0.066	0.066
HDPE film fragments			15	15
LDPE film fragments	0.48	0.96		
PA microfibres	698.6	1397.3	67.5	104.5
PA beads			0.11	0.11
PET film fragments	0.0008	2.92		
PP microfibres			242.3	370.4
PP beads			0.14	0.14
TRWP			0.000013	0.000013
Processing and packaging				
LDPE beads			35.5	35.5
LDPE film fragments			11.5	11.5
PA beads			286.6	286.6
PP microfibres	128	128		
TRWP	0.33	0.33		
Tyre wear particles for lorry transport				
TRWP	0.77	0.77	0.37	0.37
Tyre wear, passenger car				
			0.33	0.33
Mismanaged waste – fishing gear				
HDPE microfibres			0.51	1.02
LDPE film fragments	0.48	0.96		
PA microfibres	264.7	529.4		
TRWP			0.0095	0.0095
Mismanaged waste – packaging				
LDPE film fragments			19.5	39
PA film fragments			195	390
PP film fragments	17.1	34.2		

(degradation and deep burial), only at the continental scale. Degradation data specific to each polymer and generic sedimentation rates were retrieved in literature as well as in Corella-Puertas et al. (2023), while deep burial and resuspension rates were obtained in the SimpleBox4Plastics model (Quik et al. 2023).

The fate was combined with exposure and effect factors to compute CFs. For the water compartment, an exposure and effect factor (EEF) for water-dwelling species (EEFw) was first developed by Lavoie et al. (2022), later updated by Corella-Puertas et al. (2023). For sediments, Saadi et al. (2025) have developed an EEFsed, which was found to be significantly smaller than EEFw, due to the smaller exposure of sediment-dwelling species to microplastics. Both EEFs are applicable to any type of polymer, size and shape of microplastics until better data becomes available to develop more specific EEFs.

Compartmental impacts were aggregated to marine CFs by considering their dilution volumes and the ratio of species that feed in each compartment (Saadi et al. 2025). Their results show that low and medium density polymers, which have longer fate in water, greatly affect water dwelling species due to their high exposure to microplastics. On the other hand, high-density polymers were found to greatly affect sediment-dwelling species due to their fast sedimentation and small dilution volume in this compartment, even if these species are less exposed to microplastics.

The entanglement CFs from Høiberg et al. (2024) do not account for the persistence of the polymers and their lifetime in the environment, due to a lack of data on item-specific degradation in different locations and compartments. Adapting the current microplastic fate with polymer specific degradation to the macroplastic fate is something that Høiberg et al. (2024) wish to include in further work, for the purpose of harmonisation and consistency in the physical effects on biota category for plastics. Hence there is large uncertainty in scaling up microplastic fate models to use in calculating CFs for macroplastic items. Significant spatial variations are not accounted for in a static degradation model. The entanglement CFs are not polymer-specific, they should not be compared to the microplastic ones. They are modelled for a duration of five years, as the fate of the plastic gets more and more uncertain with longer modelling times (e.g., due to biofouling and fragmentation). Høiberg et al. (2024) present two possible options; one for buoyant fishing gear and one for dense polymers (e.g., nylon) expected to sink upon mismanagement. Fishing gear is both buoyant and weighted (to achieve appropriate buoyancy to be at the correct height in the water to fish the target species). The buoyant factors were used for the calculations presented in this study. Høiberg et al. (2024) state that the CF belonging to the Exclusive Economic Zones (EEZ) where the fishing activity occurs should be multiplied with the plastic losses of the specific fishing operations to arrive at the final impact. Thus, the factors from Høiberg et al. (2024) were 2.02E-11 (PAF/kg plastic, CF for the EEZ with the name "Norwegian Exclusive Economic Zone") and 1.58E-11 (PAF/kg plastic, CF for the EEZ with the name "Peruvian Exclusive Economic Zone"). These factors for Norway and Peru were multiplied with the amount of plastic in the fishing gear lost during the use phase for cod and anchoveta. The units conversion from PAF to PDF.m2.yr was performed using ocean surface area of 3.61 E + 14 m^2 and a PAF to PDF conversion factor (severity factor) of 1. The time component (yr) was included by considering the time horizon 1–5 years (see Fig. 2).

It should be noted that using data for reported losses means that unreported losses were not included.

3 Results

The loss rates used to calculate the LCI data associated with the macro- and microplastics emissions that form the LCI for the marine microplastics impacts are given in Table 2 and the LCI data in Table 4.

Figure 3 shows the ecosystem quality results for the cod and anchoveta systems. The fishing stage dominates the ecosystem quality damage results for both systems, but the dominating impacts are different. For cod fished with gillnets, the production of the antifouling coating is the dominant contributor to freshwater ecotoxicity (short term), which dominates the results for the cod system. Entanglement, biodiversity damage due to land transformation (also attributable to the antifouling coating), and freshwater ecotoxicity (long term) are the other visible contributors for the cod system. Entanglement results completely dominate the picture for the anchoveta system. In Fig. 3, a 5-year perspective has been used for entanglement effects.

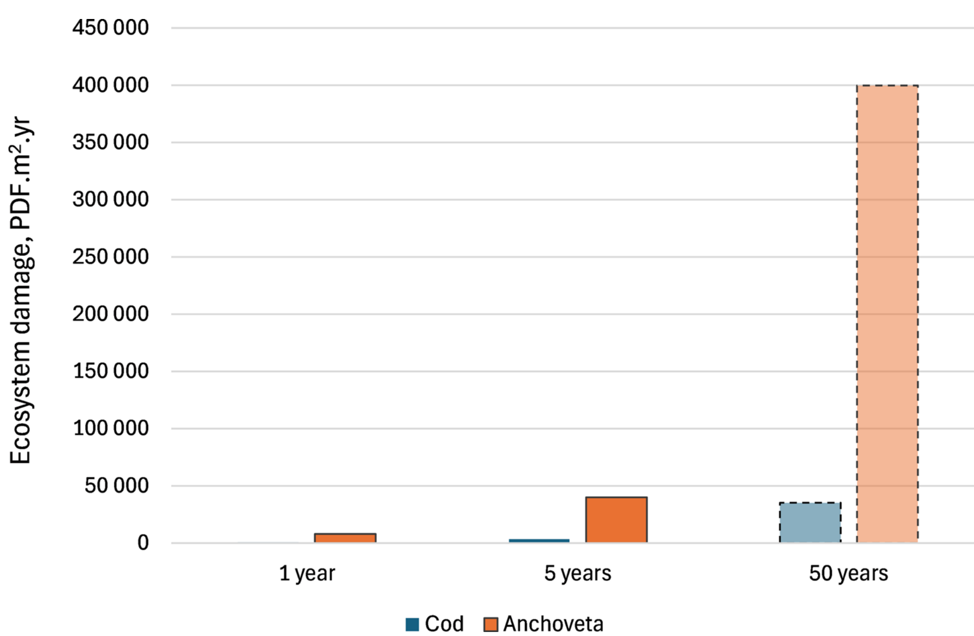

Fig. 2 Ecosystem damage results (PDF.m^2.yr) for entanglement from lost fishing gear for two fishing industry cases, cod (household consumer) and fishmeal (feed for fish farms) for the Norwegian market. The CFs used are valid for a 5 year time horizon; where the CFs are used for longer time horizons, the results are shown with dashed outlines and lighter colours

Fig. 3 Ecosystem damage results (PDF.m^2.yr) for two fishing industry cases, cod (household consumer) and fishmeal (feed for fish farms) for the Norwegian market

As mentioned in the methods section, entanglement was modelled for a duration of five years (the fate of the plastic gets more and more uncertain with longer modelling times), however some fishing gear can carry on entangling marine organisms over much longer time-scales. In order to show the importance of the time-scale, Fig. 2 shows the ecosystem damage results for entanglement given for different time-scales. One year would be the relevant time if the gear was successfully retrieved by the Directorate of Fisheries retrieval missions that happen in Norway (Vodopia et al. 2024). Five years is the time perspective covered by the CFs provided by Høiberg et al. (2024), however the longer timeframe tested, 50 years, is shown with weaker colour intensity and dashed lines to show they are much more uncertain. This figure shows that if gear should be left in the ocean, and continue to entangle marine organisms for such long time-frames, the entanglement impacts can become very large. The entanglement impacts are much larger for the anchoveta system than for the cod system. This difference is mostly due to the LCI data for the macroplastic emissions (gear losses) for the anchoveta fishery during the fishing life cycle stage.

In order to see the relative importance of the other life cycle stages, Figure 4 shows the ecosystem damage results without the fishing stage of the life cycle for both systems. Freshwater ecotoxicity (long term) is the impact category contributing with the largest overall ecosystem quality damage. The emissions contributing the most to this impact category for the cod system are emissions of aluminium, copper and iron ions to water. The contribution to this from fishing gear production is mainly associated with the lead in the bottom rope (both production and processing of the lead). The main contributor for freshwater ecotoxicity (long term) for cod processing and packaging is associated with aluminium ion emissions from polyethylene production. For the same impact category, the contribution from transport from the retailer to the household consumer for the cod system is a result of the metal ion emissions for producing the passenger car (aluminium, copper and iron). In Fig. 4, after freshwater ecotoxicity (long term) follows climate change (long term), due to emissions of fossil-derived carbon dioxide to air. Similarly, for the anchoveta fishmeal case, the impact categories contributing the most to ecosystem damage are freshwater ecotoxicity (long term) (38%), and climate change (long term) (32%). Without considering the fishing stage (Fig. 4), the production of phosphoric acid and iron sulphate used in the fishmeal reduction are the flows that contribute the most, for freshwater ecotoxicity (long term). Regarding climate change (long term), the use of fossil fuels and refrigerant (diesel, R500 and natural gas) are responsible for up to 90% of the impact. Even with 100% fragmentation into microplastics, physical effects on biota account for less than 0.1% of the total endpoint impacts for both systems.

As alternative allocation perspectives (i.e., energy and economic allocation) can affect the final results, Figure 5 shows the results for the anchoveta case study using both of these allocation approaches. The results are equally distributed in terms of impact contributions to the overall ecosystem damage, but mass allocation (used for all other results in this paper) gives the fishmeal product a larger share of the impacts than economic allocation and energy allocation. Use of energy allocation would assign the lowest impacts to the fishmeal than any of the other allocation approaches shown (Fig. 5).

Figure 6 shows the damage to ecosystem quality results from the physical effects on biota for microplastics. Production of the components that are plastic includes some waste plastic arising from the production process. Emissions of plastic from post-landing stages were from mismanaged waste of packaging and linked to indirect discharge to the ocean. Figure 5 shows the situation where 50% or 100% of macroplastic emissions to the environment fragment into microplastics.

4 Discussion

4.1 Global ecosystems damage

The results from this study show that dissipative plastic emissions from the fishing industry, particularly those linked with entanglement, could contribute significantly to the overall ecosystem quality damage, when compared to the impacts usually calculated with IMPACT World+ in LCA studies.

When the fishing stage is removed, the largest ecosystem impacts arise from freshwater ecotoxicity (long term) and climate change (long term). Microplastic physical impacts on biota are much smaller and can only be analysed correctly when examined by themselves (Fig. 6).

For the cod system, the most important process contribution to long term climate change damage is fishing (fuel burnt in fishing vessels, 29%), followed by nylon production (15%), and driving the diesel passenger car to/from the shop (7%). It should be noted that some fishing vessels can have lower marine diesel use than the literature data used in this assessment (131 l/tonne). For cod fishing in Norway, some vessels are hybrid and can have fuel consumption that is approximately 40% of the fuel consumption from the literature data (personal communication, Jørgen Vollestad). This shows that there is large room for improvement in the climate change impacts for fishing, using available technology. Given that other impacts dominate the results for the

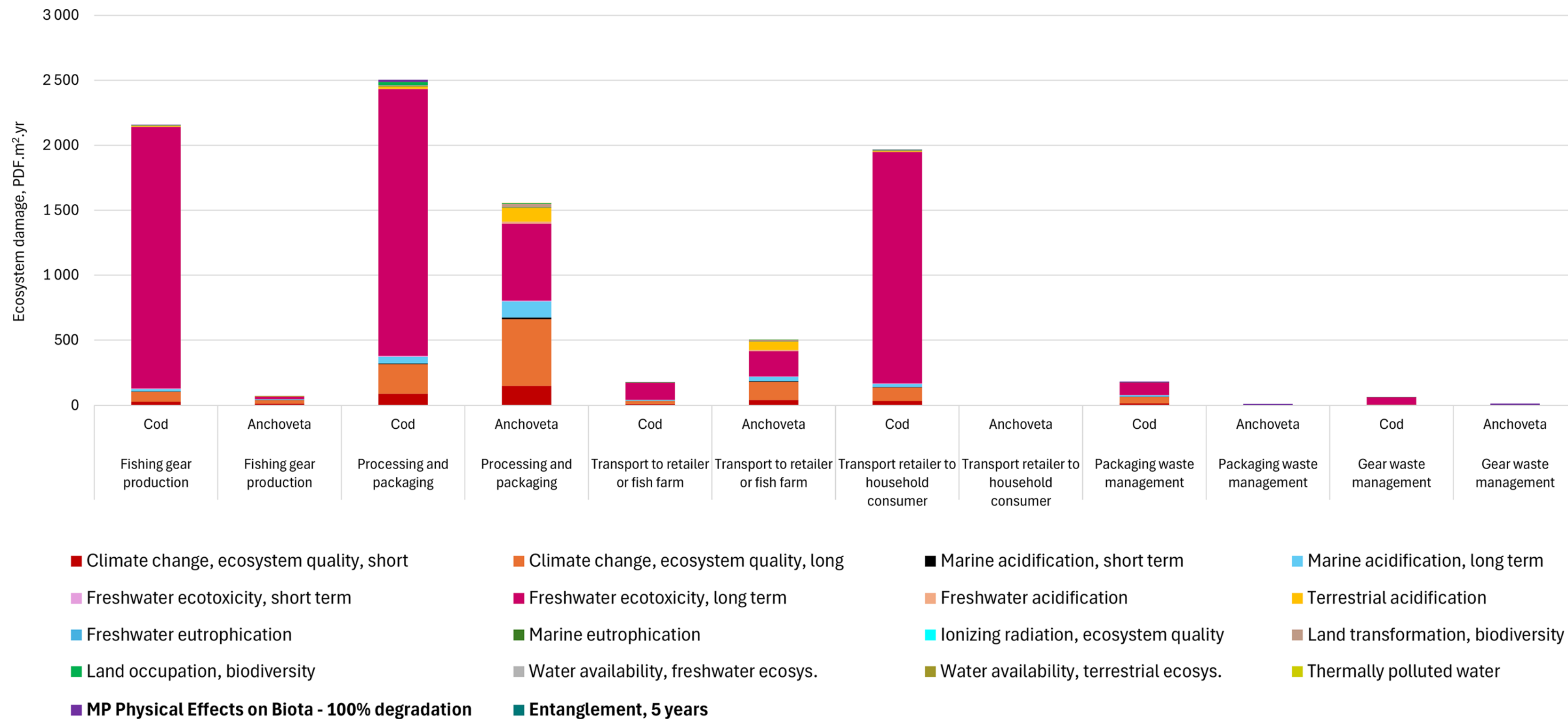

Fig. 4 Ecosystem damage results (PDF.m^2.yr) for two fishing industry cases, cod (household consumer) and fishmeal (feed for fish farms) for the Norwegian market, without the fishing stage

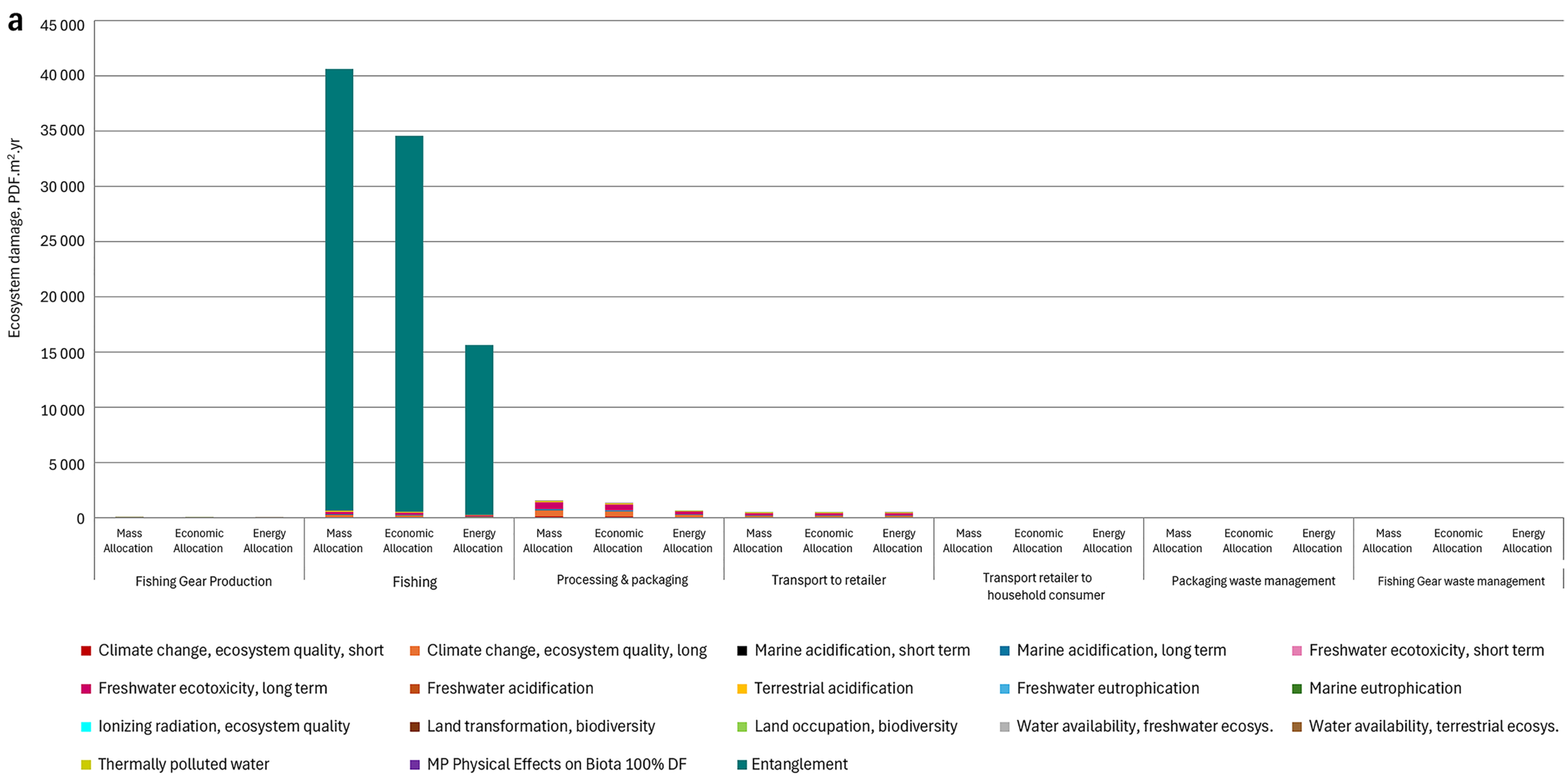

Fig. 5 Ecosystem damage results (PDF.m^2.yr) for the anchoveta fishmeal case using different allocation methods, mass, economic and energy (5.a. With entanglement and 5.b. Without entanglement)

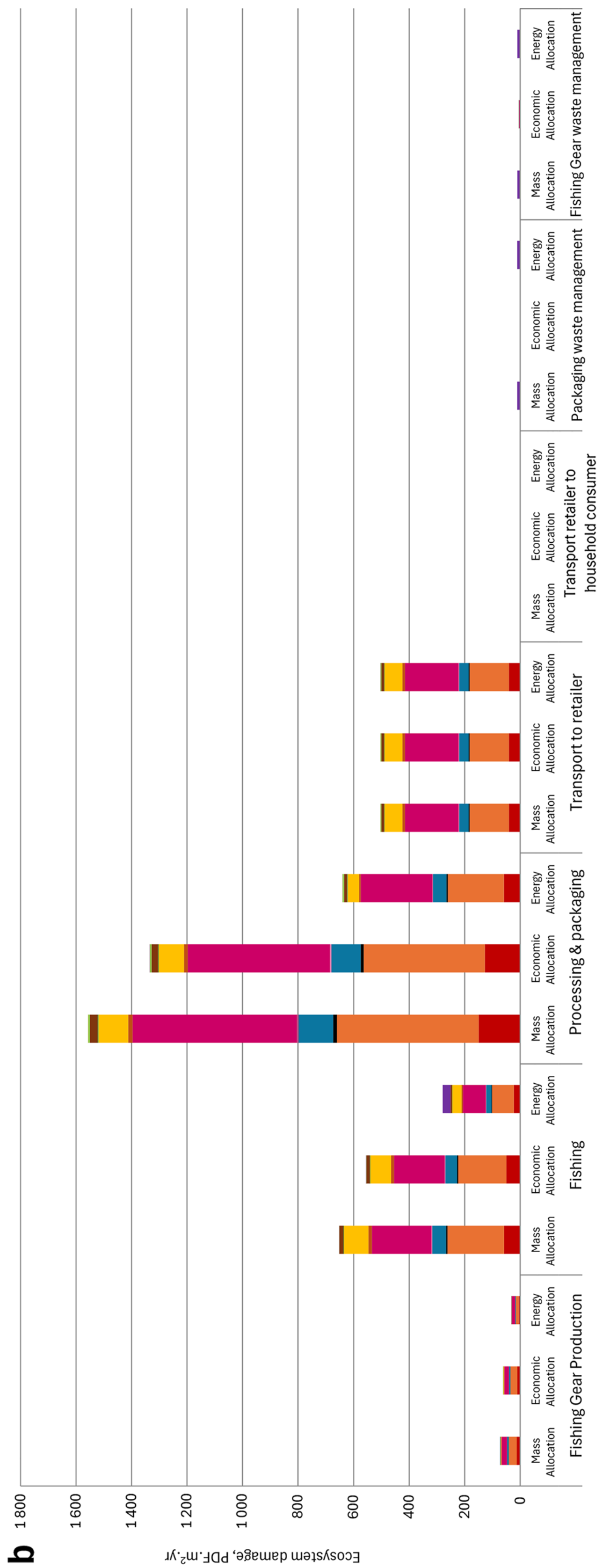

Fig 5 (continued)

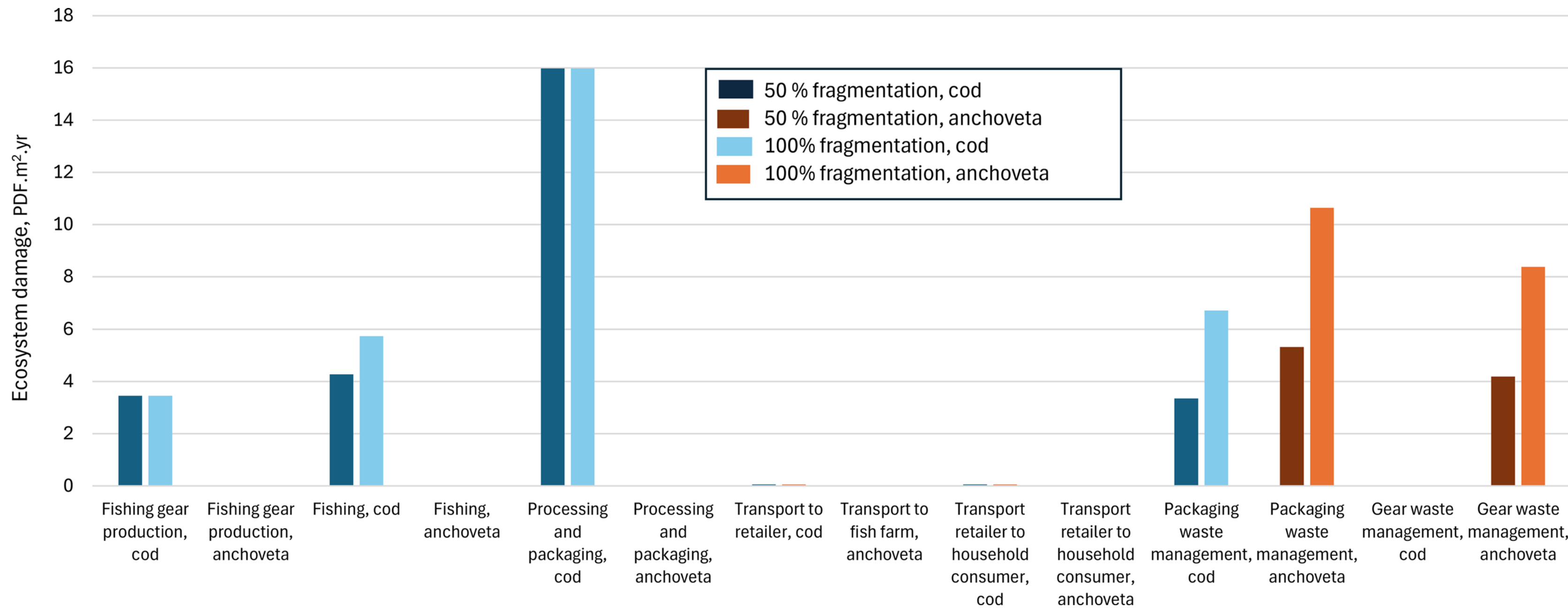

Fig. 6 Ecosystem damage results (PDF.m^2.yr) for microplastic physical effects on biota for two fishing industry cases, cod (household consumer) and fishmeal (feed for fish farms) for the Norwegian market, where 50% or 100% of the plastic emissions fragment into microplastics within a 100 year time-frame

fishing stage (Fig. 3), this variation in fuel use would not change the conclusions of this study.

The Peruvian anchoveta fishery is highly efficient in fuel use intensity, which is usually below 35 L/t (Deville et al. 2025; Avadí et al. 2014). This is substantially lower than benthic fisheries (e.g., cod) and lower than most of other pelagic fisheries reported in the literature (Parker et al. 2018). Hence, measures to mitigate climate change in the anchoveta fishery should address changes in the mix of energy carriers for vessels, rather than aiming at an optimization of fuel use.

In addition to the impacts occurring due to the life cycle activities included in both systems previously described, biotic stock depletion occurs due to the extraction of anchoveta and cod. These biotic stock depletion impacts also contribute to ecosystem quality damage. The first CFs developed to analyse the impact on the ecosystem due to the extraction of biotic stock (Langlois et al. 2014 and Helias et al. 2018) were linked to the area of protection "natural resources". Stanford-Clark et al. (2024) incorporated by-catch and linked the depletion of these biotic resources to the ecosystem quality area of protection. These methodological developments measured the impact of the extraction of the fish, based on the reproductivity capacity, but did not account for ecosystem dynamics (impacts on other species, or the ecosystem more broadly), as was needed to bring the methodology in line with the other methods used for the results calculations in this study. However, at the time of submission of this paper, Stanford-Clark et al. (2025) has just been published. In this new study their impact pathway for the target species has been "expanded to the ecosystem scale by incorporating additional indirect impacts associated with fishing through the integration of ecosystem dynamics." Thus, the authors of this paper would encourage authors of future studies to include these wild fishing impacts in future research.

4.2 Allocation

As mentioned in 2.2.1, allocation is a challenging methodological decision when considering multifunctional systems in LCA (Pelletier and Tyedmers 2011). FMFO are produced from the anchoveta system. Figure 5 shows the results for three different allocation approaches, i.e., mass, economic and energy allocation are included to produce fishmeal. These results show how the choice of allocation model can affect the final results presented for this particular case study. For the fishmeal product, the burdens allocated are larger for mass allocation than economic, the burdens allocated for fishmeal if using energy allocation are lower than for an economic allocation. Economic allocation is perceived to be a less stable allocation method (Pelletier and Tyedmers 2011), as prices can fluctuate in the market (prices between 2019 and 2024). Prices for fishmeal were about 1500 USD/t, fish oil was about 2400 USD/t (Banco Central de Reserva 2023a, 2023b). At the time of writing the fishmeal prices were about the same, but fish oil prices were much higher (up to 10,000 USD/t). In contrast, energy allocation leads to a partition of 66:34 (fishoil:fishmeal) (Fréon et al. 2017), which reduces the burdens allocated to fishmeal considerably, since the fish oil is allocated 66% of the burdens.

The changes in allocation would not change the relative importance of the fishing stage of the value chain, but the allocation approach would change the overall size of the ecosystem quality damage for fish meal.

4.3 Plastic related emissions and their impacts

The results for entanglement are dependent on the time perspective. Figure 2 shows that the environmental damage linked to entanglement depends on the time horizon that the lost gear is assumed to be able to entangle organisms. If the gear were to entangle marine organisms for a time horizon longer than five years, entanglement would dominate the damage to ecosystem quality more than it does in Fig. 3. The model used assumes that the potential for entanglement remains the same each year that is modelled. Entanglement impacts are very difficult to assess over a long time horizon, since biofouling and entanglement can affect the buoyancy of the items, and fragmentation and degradation will influence the entanglement potential. Due to ocean currents, weather conditions, biofouling and fragmentation it is impossible to confidently say where a piece of plastic will be after 50 or even 100 years, which will affect which CFs are relevant to use (e.g., in which EEZ the lost gear, or gear fragments are present). The results in Fig. 3 show entanglement modelled for a five-year time horizon, where no fragmentation is assumed. If the gear is retrieved (in retrieval missions like those described in Vodopia et al. 2024) the gear may be able to entangle marine organisms only for about a year (see Fig. 2). However, if the lost gear does not break down and still can entangle marine organisms after 50 years, then the potential ecosystem damage results for entanglement are extremely large (about 400,000 PAF.m^2.yr/tonne, see Fig. 2).

As previously described (in 2.2.3), the gear loses from the fishing stage of the life cycle are based on reported losses for the cod system. It is possible that fishers do not log all of their gear losses, even if it is obligatory to do so in Norwegian waters. This can lead to underestimation of the life cycle inventory data for gear loss. This study shows that entanglement due to loss of macroplastic is the most important environmental impact associated with this plastic loss at sea (Fig. 3). If the gear losses for gillnet fishing in

Norwegian waters were twice that reported, then the impact for entanglement in Fig. 3 would be twice as large for the cod system. This would bring the fishing impacts up to approximately 24,000 PDF.m^2.yr.

The anchoveta case includes fishers disposing of their onboard litter (like plastic bags or bottles) directly into the ocean (retrieved from Deville et al. 2023, as described in Table 2). This direct littering related to crew activities is likely to occur in a large proportion of the world's fisheries.

It is uncertain how long it will take for the macroplastic emissions from the fishing systems to break down into microplastic. The results in Figs. 3 and 4 are considering 100% fragmentation. However, conventional plastics are known to have long fragmentation times in nature (Gerritse et al. 2020). Figure 6 provides results for two different levels of fragmentation, 50 and 100%. There are a number of sources of direct microplastics emissions, e.g. wear and tear of fishing gear ropes during use, antifouling and tyre wear particles. These do not change when the macroplastic fragmentation rates change, which is why the difference between the 50 and 100% fragmentation columns is not larger than it is in Fig. 6. Figure 6 shows that fishing, packaging and production of the gear are the most important steps in the life cycle for plastic emissions that can lead to ecosystem damage due to microplastics' physical effects on biota. The damage caused by physical effects on biota from tyre wear particles from the distribution activities in the life cycle is much smaller (approximately 0.4% of the ecosystem damage due to microplastics' physical effects on biota).

Microplastics physical effects on biota cause less damage to ecosystem quality than the conventionally analysed damage categories shown in Fig. 4. The cod system includes all parts of the gear (production and losses), whereas the anchoveta fishmeal case includes only the nylon net. Production of the plastic components includes some waste plastic arising from the production process, i.e. granulate losses from plastic granulate production (Quantis and EA 2020) and discarded material due to quality control and processing losses that are sent to waste management. The mismanaged waste factors are also applied to these waste plastic flows for both cases. Emissions of plastic from post-landing stages were from mismanaged waste of packaging and linked to indirect discharge to the ocean. In fact, our inventory considers redistribution of macro- and microplastics from terrestrial and freshwater to marine environments, however, this impact category does not yet consider microplastics impacts that occur within freshwater and terrestrial compartments (Saadi et al. 2025). This likely leads to an underestimation of the impacts of plastics, that ongoing work aims at quantifying. For example, tyre wear particles are expected to be emitted into terrestrial environments and thus have physical effects on terrestrial biota.

4.4 Limitations

It should be noted that when gear is lost, the weighted ropes that are also lost contain lead. In the anchoveta fishery the lead weights are in blocks distributed evenly along the bottom ropes in the purse seine gear. For the Norwegian gillnet case the lead is woven in thinner elements through the weighted bottom ropes in the gillnet gear (e.g. lead core ropes pictured in Tateda et al. 2014). This interwoven lead is more easily broken up and ingested by marine organisms and seabirds with plastic parts of the rope. This is likely to contribute to ecotoxicity of the fishing gear. These emissions and effects are not included in this LCA study.

One of the limitations arises from the lack of plastic emissions data inclusion in databases used for LCA. The plastic waste emissions that were modelled in this study are mostly emissions from the foreground system, whereas the background system plastic emissions (e.g., particularly those associated with infrastructure and services, like electricity and water) are essentially ignored.

There are other potential impacts of plastic emissions that are still being researched and are not covered in this study. Beyond entanglement, macroplastics can lead to smothering (for example by covering coral reefs, or the roots and branches of mangrove forests; Walther and Bergmann 2022; Lamb et al. 2018) or damage through ingestion (through blockage of the digestive tract or gastric rupture; de Stephanis et al. 2013; Bråte et al. 2016; Woods et al. 2021). Ingested macro- or microplastics can release plastic additives, which can pose ecotoxic effects (Casagrande et al. 2024; Tang et al. 2022). Furthermore, plastic debris can act as a vector to transport invasive species and pathogens over long distances, potentially posing hazards to ecosystems (García-Gómez et al. 2021; Rech et al. 2018). Finally, this study focused on impacts of plastic emissions to marine ecosystems, whereas plastic debris can also pose hazards to freshwater and terrestrial ecosystems (Okoffo et al. 2021; De Souza et al. 2018; Li et al. 2018, 2023), which are so far not included in LCIA. For the fishing industry, terrestrial and freshwater plastic emissions can occur for example through losses during plastic production or mismanaged packaging waste (Loubet et al. 2022; Quantis and 2020). The inclusion of these additional impacts could further increase the importance of dissipative plastic emissions on ecosystem damage.

As LCIA models get more comprehensive, but also more complex and taxon-specific, interpretations become more challenging. Since terrestrial, marine and freshwater ecosystems are very different and differently modelled, GLAM advocates to keep results from the three realms separate. Moreover, adding them implies important choices as to what exactly is equivalent: a realm is as important as another realm, a species is as important as another species, or even an ecosystem's surface area (e.g., one square meter) has the same value as any other ecosystem's square meter.

The case studies' results have uncertainty associated with them. Uncertainty is linked to both the life cycle inventory and the impact assessment steps. Primary data were unavailable for multiple life cycle stages, making the use of secondary data necessary for the inventory (Table 1), this leads to parameter and scenario uncertainties that were not calculated in our study. In the LCIA step, model and parameter uncertainty is present across all impact categories, including the newly developed ones. For established impact categories like freshwater ecotoxicity or terrestrial acidification, the uncertainty for CFs typically spans one to two orders of magnitude (Rosenbaum et al. 2008; Roy et al. 2014). The uncertainty for physical effects on biota of microplastics is around two orders of magnitude (roughly one above and one below the mean CFs), which is in line with other impact categories (Saadi et al. 2025). The uncertainty for the entanglement CFs has not yet been estimated. While we did not run a formal uncertainty propagation or Monte Carlo (MC) analysis here, communicating uncertainty remains important. MC simulation is recommended for future work, when all impact categories have uncertainty ranges available.

This study contributes to the literature on inclusion of plastic litter impacts into LCA. However, it relies heavily on recently published CFs that are still evolving, this means that the results in absolute terms are likely to change with future implementations of updated methodology and CFs. Thus, the results presented can be considered as not strong in terms of comprehensiveness and robustness. However, the importance of the inclusion of plastic losses and littering in value chains involving plastics is shown to be important in this work. The framework presented in Woods et al. (2021) shows that there are many more types of effects that are not yet included in this current paper, so the true impacts on ecosystem services of plastic litter associated with fishing and fish products are most likely larger than those presented here.

4.5 Policy implications

4.5.1 Measures for fisheries

Annual fishing gear retrieval missions (Vodopia et al. 2024) are a measure that the Norwegian authorities have put in place to reduce ghost fishing. Since entanglement is shown to be important in this study, this is a potential measure that other countries could consider in order to reduce these impacts. More specifically, Peru has been shown to have a relatively high emission rate of plastic to the ocean, not only from ocean-sources, like fishing or cruises (Deville et al. 2023), but also from coastal littering (Vázquez-Rowe et al. 2024) and deficient municipal waste management systems (Ita-Nagy et al. 2022). Hence, the implementation of management measures for abandoned lost and discarded fishing gear (ALDFG) is necessary in Peru to mitigate one of the various sources of plastic emissions in the Eastern Central Pacific. These could include the improvement of port reception facilities for plastic waste, which remain largely unavailable in most Peruvian ports, to make it more attractive to return discarded gear, or the identification of particular areas in which fishing activities create increased emissions (e.g., areas of high current), aiming to reduce or ban fishing in those areas (Gilman et al. 2023). Moreover, if other fisheries in Peru (mainly artisanal, not included in this research) have the same control and inspections as the anchoveta industrial fisheries, the marine litter to the ocean would reduce significantly.

Requirements for ships in Norwegian law (Regulations on the delivery and receipt of waste and cargo residues from ships, 2003) mean that all fishing boats using Norwegian ports must deliver their waste to the port waste management facilities and pay the required fees to cover the costs of this waste management. Inspections are carried out by the authorities, and these include checking that waste has been delivered and fees paid as required. This law also requires the harbours to provide adequate waste management facilities. If these sorts of measures were adopted in Peru, they could contribute to improving port waste management and reducing littering on the coast and at sea.

4.5.2 Integrating plastic pollution in environmental labelling

Until recently, LCA-based environmental labelling of plastics was limited to conventional impact categories, such as climate change, water scarcity, etc. The new impact category of physical effects on biota allows for integrating potential impacts related to plastic pollution. With the recently approved EU's Product Environmental Footprint Category Rules (PEFCR) for Apparel and Footwear, the EU has made a first step towards the inclusion of plastic pollution impacts into environmental labelling. Specifically, the new PEFCR includes a mandatory component for assessing microfibre emissions and their potential physical effects on biota (using the CFs developed by MarILCA, Saadi et al. 2025). Other sectors and countries could follow this example, integrating the assessment of impacts related to plastic pollution as a mandatory component in their environmental labelling.

5 Conclusions

This study of anchoveta and cod fishing aimed to include dissipative plastic emissions in the inventory data and test currently available CFs for plastic pollution impacts to identify their importance for ecosystem quality damage.

The study provides a joint assessment of conventional and marine-specific impact categories for two fish production systems destined for Norway. The fishing stage dominates the ecosystem damage impacts for both systems. Including the impacts associated with plastic losses in the product value chain in ecosystem quality damage is important, more so for entanglement (macroplastic) than for microplastic physical effects on biota. The results presented in this paper show that the combined of these environmental categories enriches the discussion. Whether parts, or the whole of the gear (including ropes) are included affects the importance of the mismanaged waste losses in the value chain, as well as the microplastic impacts associated with fishing gear losses at sea. The timescale for when lost fishing gear can still entangle organisms is extremely important for the overall damage caused by fishing and should be investigated further.

This work shows that LCA studies that include plastic materials or products in their life cycle need to include the impacts associated with plastic losses in the LCI. Inclusion of the CFs developed for plastic impacts through projects like MarILCA will enable practitioners to include plastic effects in their EQ damage calculations. There are a range of impacts that plastics and their component chemicals can have on ecosystems, not all of these have CFs that are developed enough to include in LCA studies at the present time. However, including as many of these impacts as possible is a step in the right direction to comprehensive LCAs including all of the impacts that plastic losses (or littering) can have.

This paper adds to the body of work that provides reasons for policy-makers to put in place measures to reduce marine plastic emissions. The LCA methodologies applied in this work can support policy-makers in quantifying marine impacts and setting targets to reduce both direct plastic emissions (e.g., ALDFG) and indirect emissions through the value chain (e.g. where packaging is produced and discarded, or antifouling use and maintenance).

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Author contribution All authors contributed to the study conception and design. Mafalda Silva (MS), Cecilia Askham (CA), Alejandro Deville (AD) and Valentina Pauna (VP) contributed to modelling the two systems in SimaPro and calculating the results for the two fishing cases. VP implemented the CFs from Saadi et al. (2025) into the software for both cases. The CFs for physical effects on biota were provided by Nadim Saadi (NS). The first draft of the manuscript was written by CA and all authors reviewed and commented on and/or revised the manuscript.

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Data availability Foreground data supporting the findings of this study are available within the paper, supplementary information and references. Background data is from databases that are covered by license agreements will not be made available.

Declarations

Conflict of interest The authors have no competing interests to declare that are relevant to the content of this article.

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