Trading Responsibility: navigating national burdens in a globalized world

Navarre, N.H.

Chapter 4

The consequences of trade on global plastic pollution

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Abstract

Plastic has become an omnipresent material that negatively affects ecosystems. Many high-income nations export plastic waste to alleviate local burdens, however importing countries have less advanced plastic treatment and management infrastructures. Here, we evaluate the influence of trade on global plastic leakage to the aquatic environment by combining spatial plastic waste generation with global plastic waste trade commodity data. Plastic waste traded from high-income and mismanaged in low- and middle-income countries results in 1.2 Mt of additional plastic debris accumulating in aquatic environments, increasing previous estimates of plastic pollution from high-income countries to freshwater environments by 51% and marine environments by 100%. Improving international cooperation is essential to stop this until now underestimated but crucial source of plastic waste polluting ecosystems worldwide.

Keywords

Plastic pollution; Plastic waste trade; Waste management; Marine pollution

4.1 Introduction

Plastic waste accumulation in the environment is a threat to global ecosystems (McGlade et al., 2021). In the environment, plastics can entangle animals or birds, preventing proper respiration or complete mobility, or may be ingested inhibiting sensations of hunger leading to starvation (Li et al., 2016b). Plastics also reduce light penetration in aquatic environments and carry concentrations of harmful pathogens, degrading coral environments and affecting aquatic microbial life (Pawar et al., 2016b). In both terrestrial and aquatic systems, plastics have been found to degrade into microplastics, entering countless food chains and triggering toxic responses in exposed biota (MacLeod et al., 2021; Rodrigues et al., 2019; P. Wu et al., 2019).

To curb plastic waste leakage, countries have repeatedly coalesced and developed agreements including the 'Convention on the Prevention of Marine Pollution by Dumping of Wastes and other Matter' (The London Convention of 1972), the 'International Convention for the Prevention of Pollution from Ships 1973' (MARPOL), the 'UN Law of the Sea Convention' (UNCLOS, 1982), multiple UN resolutions spanning from 1995 to 2021, and more (Barrowclough & Birkbeck, 2022; Tessnow-von Wysocki & Le Billon, 2019). Recently, the Nordic Council of Ministers proposed a new multilateral agreement, while the United Nations Environment Assembly has signed a resolution to develop a legally binding instrument to end plastic pollution (Raubenheimer & Urho, 2020; UNEP, 2022b). One aspect largely unaddressed in these agreements, however, is a means to curb plastic waste trade. Only recently has the Basel Convention ratified an amendment preventing the export of hazardous materials, including unrecyclable plastic waste (Basel Convention COP, 2019b). Yet, with unclear definitions, lack of enforcement mechanisms, and certain major exporters refusing to sign the amendment, the output of the Basel Convention has not yielded a strong adjustment in global plastic waste trade (Khan, 2020; Wen et al., 2021).

The current plastic waste trade still shifts plastic waste from high-income countries, with low waste mismanagement rates, to low- and middle-income countries with higher rates of waste mismanagement (Barrowclough et al., 2020; Brooks et al., 2018a; C. Wang et al., 2020b; Wen et al., 2021), whereby it contaminates nearby terrestrial and aquatic environments (Chen et al., 2021; Greenpeace, 2022; UNEP, 2020; Wen et al., 2021). Unable to contain this problem, countries are increasingly banning plastic waste imports (e.g. China in 2017, Turkey in 2021; Gündoğdu & Walker, 2021; Wen et al., 2021). However, high-income countries, including many EU member states, Australia, Canada, and the United States, have largely responded by diverting their plastic waste to other nations, where volumes of imported plastic waste being dumped are surging (Khan, 2020; Sarpong, 2020). The extent to which this traded plastic waste shifts burdens and contributes to environmental plastic leakage worldwide remains unexplored, however.

Here, we develop a spatial probabilistic estimate of (traded) plastic waste that is likely to enter perennial rivers, lakes, and oceans (hereafter 'aquatic environment'). Using spatially explicit population density, national accounts of plastic mismanagement, and spatial estimates of informal plastic waste recovery we estimate the quantity of domestic plastic waste that was likely lost to the environment for 210 countries, territories, and special administrative regions (SARs) in 2019. We combine national plastic waste trade accounts of these countries in 2019, spatial port data, and national plastic mismanagement accounts to estimate mismanaged traded plastic waste. These two estimates are coupled to the aquatic environment by using high-resolution hydrography of the aquatic environment and distance-based probability functions proposed by Borrelle et al. (2020) and Meijer et al. (2021b).

4.2 Materials and methods

4.2.1 Estimating plastic waste for disposal

Country-level plastic waste generation was calculated using data reported up to 2018 by The World Bank. Their comprehensive data includes the solid waste generation of 215 countries and special administrative regions (SAR), along with the plastic fraction of the waste generated (Kaza et al., 2018). This data was used to determine country-specific plastic waste generation per capita, which we scaled to 2019 population estimates provided by WorldPop (WorldPop, 2018). For countries that did not have a reported plastic waste fraction, we used the globally averaged estimate of 10.9% (Lebreton & Andrady, 2019b).

In addition to domestically generated plastic waste, imported and exported plastic waste were included in the total plastic waste for disposal. To account for traded plastic waste, the International Trade Database (BACI; version 2) was used. This database reports balanced bilateral trade data derived from the 2019 United Nations International Trade Statistics Database (COMTRADE). The database follows the harmonized system (HS) from which the trade data of 'plastic waste, parings, and scrap' for polyethylene (PE), polystyrene (PS), polyvinyl chloride (PVC), and polymer waste not elsewhere classified could be tracked (Gaulier & Zignago, 2010).

For certain countries, the final import destination was labeled as 'Other Asia, not elsewhere specified' (OANES). To handle the ambiguity of these trade relationships, the exports of a country to OANES were redistributed to Asian countries as defined by the United Nations UN M49 area code standard (UNSD, 2022). The exports were redistributed proportionally to the current trade patterns with Asian countries of the exporter excluding OANES.

The bilateral trade data does not necessarily indicate the final import destination of the plastic waste, however (United Nations, 2016). For example, Slovenia acts as a major re-exporter of plastic waste outside Europe for other European nations, and the Marshall Islands export far more than their waste generation and reported imports (Bishop et al., 2020c; Gaulier & Zignago, 2010). To account for such unreported re-exports, we model a generation of re-exports for countries that net-export plastic waste. For these intermediary countries, a fraction of their net exports is assigned proportionally to the mass of plastic waste imported from other countries. The re-exported waste is distributed to all the net importing partner countries of the intermediary, proportionally to the current trade patterns of the intermediary, excluding the country of origin.

4.2.2 Quantifying mismanaged waste

After calculating the plastic waste generation of each country, the mismanaged waste fraction for each country was determined. National mismanaged waste fractions were calculated in two steps. First, mismanaged waste rates per disposal pathways and income level were established following the definitions provided by The World Bank and scientific literature (Borrelle et al., 2020; Jambeck et al., 2015b; Kaza et al., 2018; Lebreton & Andrady, 2019b). The mismanaged waste rates were then applied to the plastic waste fractions entering each disposal pathway based on national waste management data provided by The World Bank (Kaza et al., 2018). Regarding traded plastics, investigations by Chinese authorities revealed that 60% of firms importing plastic waste were in violation of waste processing activities, with many of these firms establishing facilities in other countries after the China import ban (Sarpong, 2020; Tan et al., 2018). Other research has indicated that plastic waste imports are frequently mixed with domestic waste streams in Asia, while research conducted in South American ports indicated plastic waste imports are not preferentially recycled over domestically generated plastic wastes (Gobbi et al., 2019; Liang et al., 2021). Further work has found that in many countries only a small percentage of imported plastic waste is recycled, with dumpsites observing an increase of imported plastic waste being dumped or burned (Sarpong, 2020). In certain cases, unrecyclable mixed plastic waste may be mislabeled to circumvent the trade regulations established by the Basel Convention, while in other situations, the receiving party may lack the necessary infrastructure to process the imports, raising the probability of plastic waste mismanagement (Basel Convention COP, 2023; Gündoğdu & Walker, 2021; Khan, 2020; Sarpong, 2020; UNEP, 2020). We therefore assumed imported waste would follow similar disposal pathways to domestic waste. Nevertheless, since a significant portion of this waste is classified as exported to be recycled, we explore a range of mismanagement rates for imported plastic waste relative to domestic mismanagement rates (see sensitivity analysis).

The World Bank reports waste disposal pathways for 163 countries and SARs (Figure 4.1A, Supplementary Table 1). To supplement the World Bank data, we utilized estimates from Lebreton and Andrady (2019; Figure 4.1B), who relied on data from the World Atlas (Lebreton & Andrady, 2019b). Using two different sources of mismanaged waste rates provided a range of values for each country, reflected in the ranges presented in our results. The minimum and maximum results were developed using the minimum and maximum mismanagement rates provided for each country, while the midpoint results were calculated using the mean value of the two data sources.

For the countries and SARs that had no reported waste disposal pathways, a K-nearest neighbor (KNN) regression was performed. A non-parametric regression tool was used as the data showcased multiple outliers and a Shapiro-Wilk test revealed the data did not follow a normal distribution. The KNN regression was necessary for 38 countries and SARs (19%) using the World Bank dataset and 47 (23%) using the dataset compiled by Lebreton and Andrady (2019). Gross domestic product (GDP) per capita of each country for a base year of 2016 (constant 2011 international US$) and reported mismanaged waste fractions were used as the training data to determine the mismanaged waste fraction of countries with no reported data (Figure 4.1). We utilized GDP as the independent variable, weighing points inversely to neighbor distance, to train the model due to its statistically significant relationship with mismanaged waste found in previous research (Lebreton & Andrady, 2019b). To account for waste that was littered before reaching the plastic waste network of countries, a 2% litter rate was added to the mismanagement rate of all countries, capped at a maximum of 100% (Borrelle et al., 2020; Jambeck et al., 2015b).

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Per capita domestic mismanaged plastic waste $DM_{c,cap}$ for the year 2019 was calculated for each country c using the following equations:

$$L_c = P_c * l_c \tag{4.2}$$

$$DM_{c,cap} = \frac{L_c + (P_c - L_c - E_c) * f_c}{Pop_c} \tag{4.3}$$

Where L is the littered waste, P is the plastic waste generated, l is the litter rate, E is the exported plastic waste, f is the fraction of mismanaged waste in addition to littering, and Pop is the 2019 population of country c.

Domestically generated mismanaged plastic waste of each country was mapped to 2019 population spatial datasets at 30 arcseconds resolution (~1 km at the Equator; Figure 4.2A) (WorldPop, 2018). We assumed that waste generation and management was uniform across each country, mapping directly to the spatial population distribution:

$$DM_i = DM_{c,cap} * Pop_i \tag{4.4}$$

Where DM_i is the domestically generated mismanaged plastic waste found in an any cell i containing a population Pop_i.

To spatially map mismanaged imported plastic waste, we collected the geographic location of all ports from the World Port Index and filtered these results based on certain parameters (World Port Index, 2019). First, only ports which accepted international goods were considered for this analysis. Further, we assumed that the two largest classifications of ports found in a country (large, medium, small, very small) would account for the totality of plastic waste imports. Further, we assumed each port meeting these criteria would import the same fraction of national imports. The mismanaged fraction of imported plastic for each country c and port p was calculated using the following equation:

$$IM_p = \sum_p I_p * f_c \tag{4.5}$$

where IM_p is the mismanaged imported waste of port p, I is the plastic waste imported per port p, and f is the fraction of mismanaged waste excluding littering of country.

Due to the uncertainty in estimating the precise location at which imported plastic waste may be mismanaged, we spatially disaggregated the predicted mismanaged plastic waste fraction uniformly across a 50km radius buffer around each port (or to the nearest international border or coastline if closer than 50km). This method implies that if waste is mismanaged from ports, it will happen within 50km of the port to minimize transport costs (Law et al., 2020b).

$$IM_i = \frac{IM_p}{A_{\beta,50}} * A_i \tag{4.6}$$

Where IM_i is the imported mismanaged waste in a cell i, $A_{\beta,50}$ is the area of the 50km radius buffer, and A_i is the area of cell i within the buffer area (Figure 4.2B).

4.2.3 Accounting for informal plastic waste recovery

The informal recycling sector plays a critical role in plastic waste management, particularly in non-high-income countries (Kumar et al., 2018). To account for the role this system has in retrieving and valorizing potentially mismanaged plastic waste, we utilized the estimates put forth by Lau et al. (2020) to determine a population of 'waste pickers' in peri-urban to city environments and their annual collection rates (Florczyk et al., 2019; Lau et al., 2020).

From these parameters, a spatial population of waste pickers was developed, combining the fraction of waste pickers in urban environments with spatial population datasets. Mismanaged waste recovery from waste picking was then determined based on this population mask and the annual collection rate of the waste pickers (Figure 4.2C). In cells where both traded and domestically mismanaged plastic was found, we assumed that the fraction of waste recovered would be proportional the fraction of domestic and traded plastic waste modeled in the cell.

4.2.4 Calculating mismanaged waste that enters the aquatic environment

The spatially mismanaged plastic waste was then connected to the aquatic environment. We defined the aquatic environment as any cell traversed by a perennial river, lake (larger than 100 km^2), or ocean. Following to the definition given by Messager et al. (2021), we defined a perennial river as any river or stream that had less than a 50% likelihood of flow cessation of at least one day per year (Messager et al., 2021). The global river network dataset, filtered to only include rivers meeting this definition, was overlayed with our spatially projected mismanaged waste. To determine the likelihood of plastic waste entering the aquatic environment, we followed the principles established by Borrelle et al. (2020). Their work linked the probability of plastic waste entering the aquatic environment to the distance of the waste from the aquatic environment using the following equation (from Borrelle et al. (2020):

$$\alpha_i = 1 - U \log_{101}(d_i + 1) \tag{4.7}$$

Where α_i is the probability that mismanaged waste originating in any cell i will enter the aquatic environment from a hydrologic flow distance to the aquatic environment of d_i; U is a random variable drawn from a uniform distribution between 0.9 and 1 (Borrelle et al., 2020). The equation assumes that plastic mismanaged in a cell containing a perennial river will enter the aquatic environment, while mismanaged plastic more than 100 km away from the aquatic environment will have a near 0% chance (Borrelle et al., 2020). If the flow path of a plastic debris traversed a non-aquatic inland sink, it was assumed the plastic debris would not enter the aquatic environment (Figure 4.2D).

Mismanaged plastic waste that enters the aquatic environment was calculated as the sum of net domestic and imported mismanaged waste found in all cells such that:

$$E_{aq} = \sum_i \alpha_i * (DM_i + IM_i - PW_i) \tag{4.8}$$

Where E_{aq} is the aquatic environment, DM_i is the domestic mismanaged plastic waste, IM_i is the imported mismanaged plastic waste, and PW_i is the picked plastic waste in any cell i.

To build national contribution accounts, equation (4.8) was used, summing all cells found in a country. To calculate the added responsibility AR of a country c to plastic debris in the aquatic environment from trade, we used the following equation:

$$AR_c = \sum_{imp} E_{aq,IM,imp} * f_{imp,c} \tag{4.9}$$

where $E_{aq,IM,imp}$ is the imported plastic waste entering the aquatic environment of importer imp and $f_{imp,c}$ is the fraction of imports to importer imp attributed to country c.

3.2.5 Modeling plastic waste flows to oceans

To model the flux of aquatic plastic debris to oceans we used the methodology proposed by Meijer et al. (2021). Their work determined the probability $P(O)$ of plastic waste being transported from a river to the ocean using the following equation:

$$P(O) = \left(\frac{\sum_{i=1}^{n}(\theta * SO_\iota + \iota) * (\kappa * Q_i + \mu)}{n}\right)^{D_{River}} \tag{4.10}$$

Where θ is a probability coefficient related to the Strahler stream order (SO_ι) of the cell i and ι is a minimum threshold. κ is a probability coefficient relative to the river discharge (Q_i) of cell i and μ is a minimum threshold coefficient. Finally, D_{River} is the distance to the ocean of cell i and n is the number of cells from the river entry point to the ocean (Meijer et al., 2021b). In addition, we accounted for dams and reservoirs, which we assumed would act as sinks of plastic waste, preventing plastic from flowing further downstream (Lebreton et al., 2017). To do so we used a spatial dataset of global dams and reservoirs and set $P(O)$ to zero for all river cells upstream of dams or reservoirs (Zhang & Gu, 2023). We acknowledge however that dams are unlikely to entirely prevent plastics from flowing downstream, making our estimate of plastic from rivers to oceans conservative.

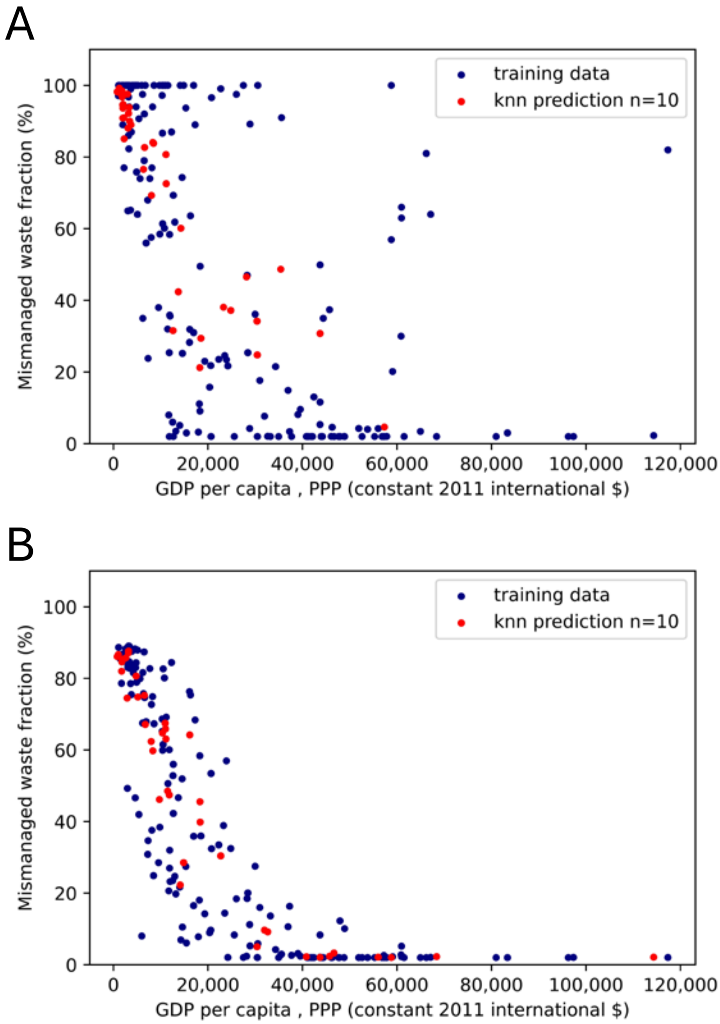

Figure 4.1 Mismanaged plastic waste fraction, including littering, for each country reported by **(A)** The World Bank and **(B)** Lebreton & Andrady (2019), overlaid with mismanaged waste fraction predicted based on gross domestic product per capita (n neighbors = 10) for countries without any reported disposal pathways.

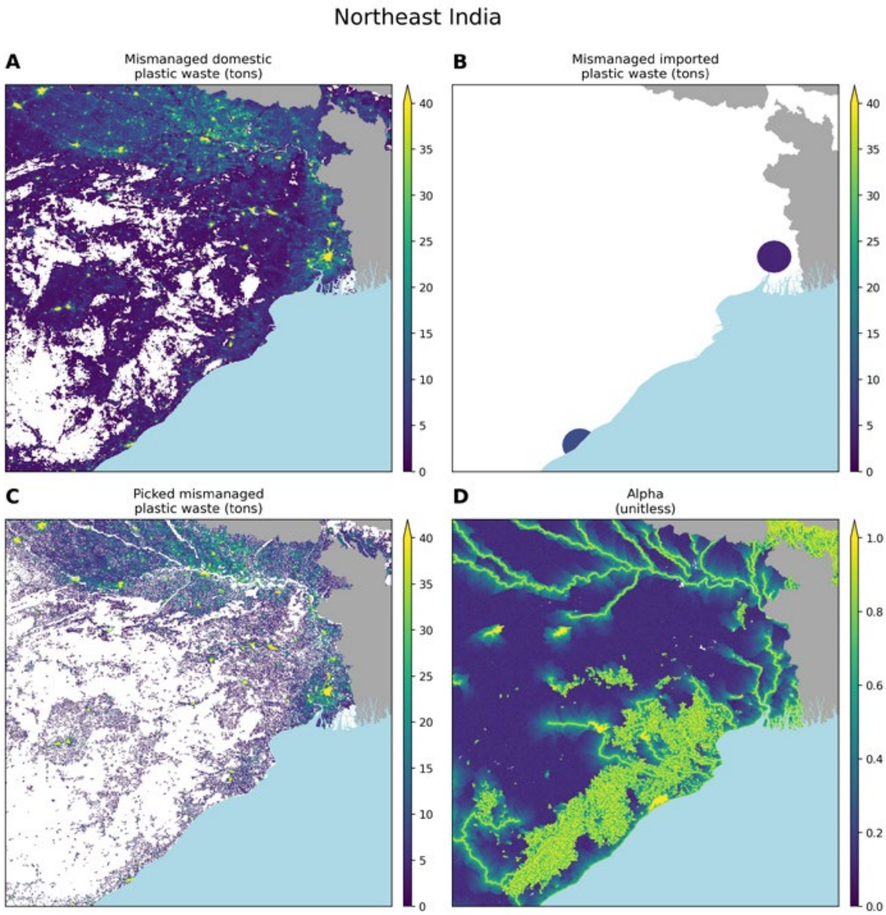

Figure 4.2 To calculate mismanaged plastic waste that enters the aquatic environment, mismanaged plastic waste is estimated spatially using population **(A)** and port **(B)** distributions, from which a fraction of this waste is re-collected by the informal waste management sector **(C)**. The probability that the remaining mismanaged plastic waste enters the aquatic environment is determined based on the alpha coefficient **(D)**, calculated using Eq. (4.7), assigned to the location of the remaining mismanaged waste, where 1 is the aquatic environment and 0 indicates no possibility of entering the aquatic environment. The figure is zoomed on Northeast India for illustrative purposes, all global datasets utilized in the analysis can be found at https://zenodo.org/records/8276941.

4.3 Results

As a basis for modeling plastic leakage of traded plastic waste, we first quantify the amount of domestic plastic waste in 2019 which was generated (235 Mt) and mismanaged (92 Mt) globally. From this domestic mismanaged plastic waste, we model that 34 Mt (22-48 Mt) entered the aquatic environment, of which 14 Mt (9-18 Mt) directly entered coastal environments (i.e. aquatic environments within 50 km of a coastline; Figure 4.3). Finally, using a spatially probabilistic model of plastic fluxes from rivers to oceans, we estimate that 1.7 Mt (1.1-2.2 Mt) of domestically mismanaged plastic waste flowed into oceans in 2019. The supplementary results (Appendix C) compare our findings with previous modeling exercises and find they are in good agreement (Borrelle et al., 2020; Jambeck et al., 2015b; Lau et al., 2020; Lebreton & Andrady, 2019b; Lebreton et al., 2017; Meijer et al., 2021b; Schmidt et al., 2017).

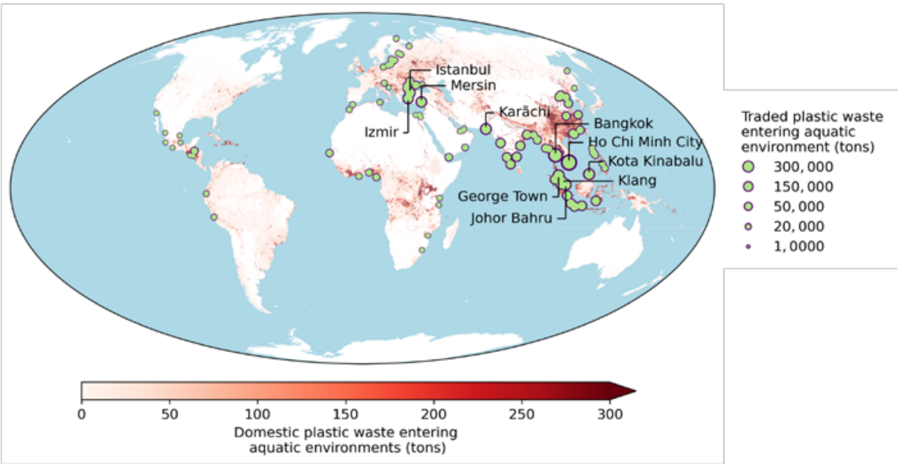

Figure 4.3 Point source emissions of domestic plastic waste (aggregated to level 12 HydroBASINS resolution) calculated to enter the aquatic environment in 2019. Traded plastic waste emissions to the aquatic environment are overlayed in green. The ten ports generating the most aquatic plastic debris worldwide are labeled (full dataset in Appendix C; Table C4).

4.3.1 The contribution of traded plastic waste

In 2019, 8.2 Mt of plastic waste were traded representing 3.5% of global plastic waste generation (Appendix C; Table C1). Of this traded plastic waste, 6.8 Mt was exported from high-income countries, and 4.4 Mt was imported by low- and middle-income countries shifting waste from countries with an average mismanagement rate of 4% to countries with an average mismanagement rate of 58% (Appendix C; Table C1).

These trade patterns led to 1.5 Mt (1.1-1.9 Mt) of all traded plastics being leaked in the aquatic environment in 2019 (Figure 4.3, Appendix C; Table C2). Based on our assessments, traded plastic waste accounts for more than 10% of direct emissions to coastal areas, adding an additional 1.4 Mt (1.0-1.8 Mt) of plastic debris and 0.17 Mt (0.13-0.22 Mt) in oceans, making traded plastics the third largest source of ocean plastics after Indonesia (0.71 Mt; 0.77-0.64 Mt) and China (0.28 Mt; 0.15-0.41) (Appendix C; Table C2). Traded plastics thus represent 10% of ocean plastic pollution despite only accounting for 3.5% of the plastic waste generated.

The traded plastics are primarily leaked around the ports of Ho Chi Minh City, Bangkok, Johor, Karachi, and Istanbul (Figure 4.3). We also highlight the port of Izmir among the largest sources of aquatic plastic debris from traded waste, a port which has previously faced criminal charges for mismanaging imported plastic waste (Gündoğdu & Walker, 2021). Trade therefore creates additional pressures on port basins with high waste mismanagement rates by concentrating the plastic waste generation from much larger areas in higher-income countries. These trade patterns allow certain nations to reduce their local riverine plastic pollution by using exports to circumvent their domestic plastic waste networks (e.g. exports from Italy to Turkey), at the cost of transferring this waste to the coastal and marine environments of other countries (Figure 4.3, Appendix C; Table C2).

Our model also calculates the direct emissions of plastics to oceans by country, and their indirect emissions via plastic waste trade. We find that for 16 countries trade more than doubled (increase of more than 100%) their emissions of plastic debris into the aquatic environment, while another 9 increased their total leakage by more than 50%. Of these 25 countries, 23 were found to be high-income countries (Table 4., full dataset in Appendix C; Table C2). Overall, these 25 countries are estimated to export 1.7 kg/cap of plastic waste that will be leaked to the aquatic environment, representing a 185% increase from estimates only accounting for domestically managed plastic.

Table 4.1 Estimates of countries and SARs found to increase their per capita plastic waste leakage to the aquatic environment by more than 50% when accounting for trade. Domestic: mismanaged domestically generated waste; imported: mismanaged imported waste; exported: mismanaged exported waste (i.e., mismanaged in the recipient country); Δ trade: difference between exported and imported; increase (%): percentage increase relative to domestic. The full dataset is available in Appendix C; Table C2.

		Aquatic plastic debris (kg/cap)				
Country Name	**Income group**	**Domestic**	**Imported**	**Exported**	**Δ Trade**	**Increase (%)**
Marshall Islands	UMC	0.04	0	571.8	571.8	1,435,550
Hong Kong SAR, China	HIC	2.6	0.0	9.4	9.4	364
Belgium	HIC	2.2	0.2	6.3	6.1	280
Malta	HIC	3.2	0.0	3.4	3.4	104
Netherlands	HIC	1.1	0.0	2.9	2.9	258
Japan	HIC	0.6	0.0	2.8	2.8	487
Cyprus	HIC	2.6	0.0	2.3	2.3	88
New Zealand	HIC	0.8	0.0	2.2	2.2	267
Australia	HIC	0.3	0.0	2.2	2.2	618
Slovenia	HIC	1.5	1.3	3.4	2.1	142
Singapore	HIC	0.4	0.0	1.7	1.7	440
Norway	HIC	0.3	0.0	1.4	1.4	528
United Kingdom	HIC	2.4	0.0	1.4	1.4	56
Sweden	HIC	0.4	0.0	1.1	1.1	254
Austria	HIC	1.5	0.0	1.0	1.0	69
Denmark	HIC	0.2	0.0	0.9	0.9	545
Iceland	HIC	1.0	0.0	0.9	0.9	95
Costa Rica	UMC	1.3	0.1	0.9	0.8	62
Czech Republic	HIC	0.1	0.0	0.6	0.6	429
Portugal	HIC	0.7	0.0	0.5	0.5	79
France	HIC	0.8	0.0	0.5	0.5	67
Spain	HIC	0.4	0.0	0.5	0.5	103
Finland	HIC	0.1	0.0	0.4	0.4	415

Saudi Arabia	HIC	0.6	0.0	0.3	0.3	57
Canada	HIC	0.3	0.0	0.3	0.3	97

In total, high-income countries increase their share of plastic leakage to the aquatic environment by 1.2 Mt (or 51%) when accounting for the leakage of their exported plastic waste. This increase is largely led by exports from Japan, Germany, the United States, and the United Kingdom (Figure 4.4). Notably, the Marshall Islands export a disproportionate amount of plastic waste relative to their domestic production and reported imports (Table 4.1; Liang et al., 2021). We presume the country likely plays a crucial role in re-exporting unreported or illegal plastic waste from larger high-income waste generators.

The waste exported from these high-income countries is transferred to 60 countries that clearly show a decrease in their share of plastic waste to aquatic environments when excluding trade (Figure 4.4). However, the majority of these exports are concentrated to a small number of major importers, namely Malaysia, Vietnam, Indonesia, Thailand, and India who are estimated to leak more than 1.0 Mt of plastic waste generated abroad in their aquatic environments (Figure 4.4), and 0.13 Mt to their oceans (Appendix C; Table C2).

Focusing on plastic leakage to oceans, high-income countries are estimated to generate 0.13 Mt of marine plastic debris in 2019 when only accounting for domestically treated waste (Figure 4.3, Appendix C; Table C2). When incorporating their traded waste accounts, in addition to domestic accounts, high-income countries are estimated to be responsible for 0.27 Mt, doubling previous expectations (

Table 4.2). The addition of traded accounts leads to high-income countries being responsible for 15% of the 1.7 Mt of plastic pollution leaked to marine environments worldwide in 2019. Since most high-income countries have little to no waste mismanagement (Jambeck et al., 2015b; Lebreton & Andrady, 2019b), plastic pollution has been framed as a societal issue in these countries, focusing on changing consumption habits and littering behavior; however, we highlight the importance of national policies regarding the export of plastic waste as an equally critical source of marine pollution which must be addressed in any future plastic pollution prevention agreements (González-Fernández et al., 2021).

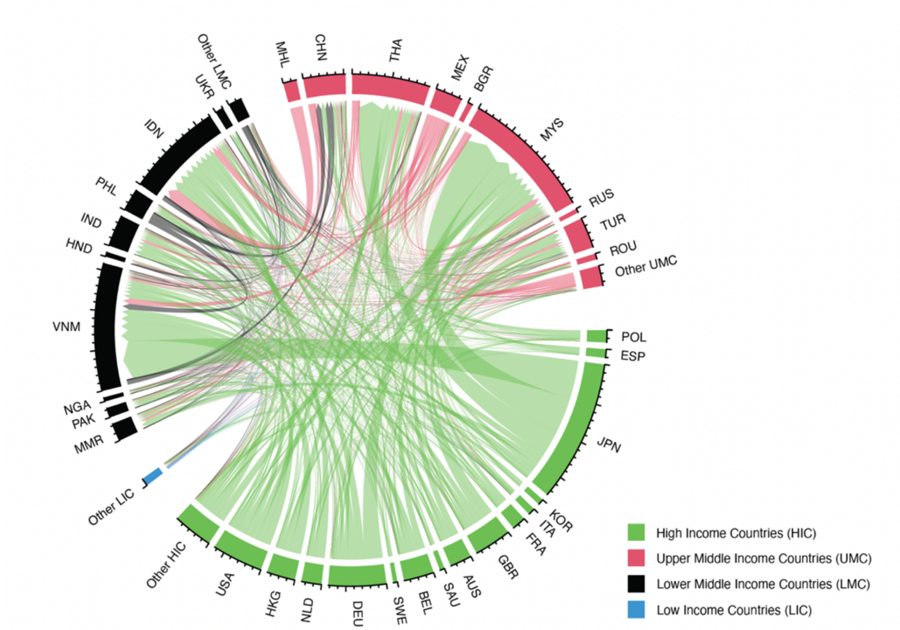

Figure 4.4 Global trade of plastic waste leaked to the aquatic environment in 2019 (major tick marks = 100 kt). Flow directionality indicates waste generated in a country that is leaked to the aquatic environment of the importing country. Individual countries importing or exporting less than 10 kt are aggregated into 'Other' of their respective income group. Full dataset identifying country ISO3 codes and income groups available in Appendix C; Table C3.

Table 4.2 Ten largest high-income countries exporters of ocean plastic pollution by total mass in 2019. Domestic: mismanaged domestically generated waste; imported: mismanaged imported waste; exported: mismanaged exported waste (i.e., mismanaged in the recipient country). The full dataset is available in Appendix C; Table C2.

	Marine plastic debris (tons)		
Country Name	**Domestic**	**Imported**	**Exported**
Japan	13795	0	39862
Germany	5751	928	17376
United States	15773	0	13482
United Kingdom	14912	0	10085
Belgium	1207	245	8027
Australia	918	0	7175
Hong Kong SAR, China	2842	16	6875
Netherlands	2122	4	5577
France	2098	1	4191
Poland	1213	691	3853
Rest of HICs*	73374	2006	22226
Total	**134006**	**3890**	**138729**

*High-income countries

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4.4 Discussion

Our analysis reveals that high-income countries underestimate their contributions to aquatic plastic debris by 51% and to ocean plastic litter by 100%. With trade from high-income countries to low- and middle-income countries foreseen to grow 50% by 2040, the fraction of plastic ocean debris from waste generated in high-income countries will continue to grow (Lau et al., 2020). For this reason, we stress the disconnect between the policy ambitions of high-income countries to curb plastic pollution and their reliance on exports as a disposal method of plastic waste. For instance, the multilateral agreement to prevent plastic pollution recently proposed by the Nordic Council of Ministers only briefly addresses plastic waste trade despite exports being the primary plastic waste disposal method of multiple Nordic countries (Appendix C; Supplementary Table 1). The High Ambition Coalition To

End Plastic Pollution, largely composed of high-income countries and lacking representation from major importers in South and Southeast Asia, also makes little mention of the plastic waste trade in their outline of a legally binding instrument (*HAC Homepage*, n.d.). Many of these agreements defer discussions of trade to the Basel Convention amendment, ignoring its limitations regarding human and financial resources to be consistently enforced (Simon et al., 2021; C. Wang et al., 2020b). With the exception of the Basel Convention, the plastic waste trade policy space has largely been dominated by import bans from low- and middle-income countries (e.g. China, Malaysia, Vietnam, and more) which have been hindered by a lack of financial resources, disorganization and corruption (Barrowclough & Birkbeck, 2022; Sarpong, 2020). As a result, despite repeated attempts to establish policy instruments limiting plastic pollution, the United Nations Environment Programme determined that the vast majority of these have been ineffective, citing a lack of standards in monitoring and reporting of traded plastic waste, among other challenges (Raubenheimer et al., 2017; Simon et al., 2021). We encourage all future negotiations to discuss and directly address plastic waste trade, highlighting the importance of putting the onus of monitoring traded plastic waste on exporters rather than importers. We also highlight the importance of addressing this aspect of the plastic life cycle in physical terms, rather than monetary terms, to properly reflect the scale of this issue and its impact on the environment.

To date, high-income countries' plastic waste policies has largely centered around maximizing recycling rates to improve the circularity of plastics (Wagner & Schlummer, 2020). Critically however, high-income countries typically incorporate exported plastic waste within their recycling metrics (Bishop et al., 2020c; Damgacioglu et al., 2020; Di et al., 2021; Hossain et al., 2022; Ishimura, 2022; Velis, 2015). Particularly with respect to European countries, previous research found that nearly half of post-consumer plastic waste reported as recycled is instead exported (Bishop et al., 2020c; Velis, 2015). Such conditions lead to high-income countries artificially increasing their plastic waste recycling rates to meet policy ambitions at the cost of relying on exports and shifting the burden of waste management to low- and middle-income countries (Figure 4.4). Our model estimates that 33% of European plastic waste collected for recycling in 2019 was exported, in line with previous estimates, however we highlight certain countries potentially export all of their plastic waste collected for recycling (Appendix C; Supplementary Table 1) (Bishop et al., 2020c). The discrepancy we expect between reported recycling rates, which incorporate exports, and true recycling rates, puts into question the environmental performance of these policy decisions (Bishop et al., 2020c). Impact assessment tools used to develop these policies do not account for the 1.5 Mts of exported and leaked plastics and do not contain adequate impact categories to represent the environmental burdens of plastic waste leaked to the environment (Boulay et al., 2021). Instead, these flows of plastic waste are evaluated entirely as recycled, significantly underestimating their environmental burden, while overestimating the capabilities of high-income countries to manage plastic waste (Bishop et al., 2020c; Boulay et al., 2021; Wen et al., 2021).

We note that our results may underestimate the contributions of trade plastics to global plastic pollution. Unreported or falsely labeled plastic waste continuously plagues the plastic waste trade, while other HS codes may include plastics, such as man-made textiles (Khan, 2020; Sarpong, 2020). Although some of our results indicate the occurrence of this phenomenon, more transparent labeling of waste and records of re-exports would improve these estimates. Further uncertainties arise from inconsistencies in reported plastic waste generation and national waste management (Kaza et al., 2018; Lebreton & Andrady, 2019b). The current work does not include microplastics; which although not expected to contribute significantly to total mass estimates, may disproportionally contribute to the toxicity and environmental impacts of plastics in various ecosystems (Julienne et al., 2019). The contribution of marine debris from fishing gear was also not included in this study, which may increase marine plastic pollution estimates by 20% or more (Isobe & Iwasaki, 2022; Morales-Caselles et al., 2021). Further uncertainties surrounding the use of probabilistic models to map the fluxes of plastic waste from terrestrial to freshwater and ultimately marine environments are discussed in the supplementary results and sensitivity analysis (Appendix C).

4.5 Conclusions

High-income countries should acknowledge that exporting vast quantities of plastic waste to low- and middle-income countries implausibly inflates recycling metrics and significantly underestimates the environmental burden of their waste. These patterns also place undue pressure on major importing nations, forcing them to implement bans that require significant political coordination, human capital, and financial resources (Sarpong, 2020). In recognizing the limitations of current plastic impact assessment tools which encourage recycling at all costs, high-income countries may better understand the importance of moving away from increasing recycling rates through exports. Instead, we encourage these countries to increase domestic recycling capacity or to consider alternative plastic waste disposal pathways that are traceable and verifiably do not contribute to plastic pollution. In addition to more transparent disposal methods, continued policies addressing the quantities of plastic waste generated and their recyclability will also be crucial to curb plastic pollution (Borrelle et al., 2020). Our results highlight that the issue of aquatic and marine plastic pollution from high-income countries stems from more than consumption and behavioral issues, but also national policy decisions. We stress that future proposed binding agreements to prevent plastic pollution will continue to be ineffective if they only acknowledge the former and if high-income countries do not directly acknowledge their contribution to this problem as a result of their trade patterns.

4.6 CRediT authorship contribution statement

Nicolas Navarre: Conceptualization, Methodology, Data curation; Writing – original draft. **José Mogollón**: Conceptualization, Writing – review & editing. **Arnold Tukker**: Conceptualization, Writing – review & editing. **Valerio Barbarossa** Conceptualization, Writing – review & editing, Validation.

4.7 Acknowledgements

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