

Abundance, Morphology and Diversity of Microplastics on Beaches in Northern Ireland

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Abstract: Microplastics pose a growing environmental threat across beaches globally. A first step in management is to assess their concentration in the natural environment and identify sources and transportation vectors. In this study microplastic abundance, morphologies, colourations, diversity and relationships with potential influencing variables were assessed for six different beaches along Northern Ireland's north coast. Beaches with a range of anthropogenic pressures were sampled in summer and winter to evaluate sources and transportation mechanisms of microplastics. Significant differences were noted in microplastic abundances between urban and rural beaches ($p=0.004$), and between summer and winter ($p=0.03$). Fibres were the most abundant morphology (76.6%) and blue was the most common microplastic colouration (46.7%). Blue fibres were the most observed morphology/colouration combination (40.7%). Fibres correlated positively with river presence and storm water outlets, while fragments correlated with urbanisation. Microplastics were ubiquitous across all beaches regardless of beach rurality or season. Pollution risk (higher MP abundances and diversity) increased in summer and on urban beaches. Summer tourism, rivers and storm water outlets were identified as likely transportation vectors.

1 Introduction

Global plastic production has been rising since its introduction in the 1950's (Hale *et al.* 2020). Due to varying industrial uses, microplastics (MPs) exhibit an array of different colours and morphologies (i.e., fibres, fragments, foams, pellets and films) and chemical compositions. First coined by Thompson *et al.* (2004), MPs are synthetic polymers defined as being ≤ 5 mm (Huang *et al.* 2023) and can be placed into two categories. Primary MP, which are intentionally created for commercial use, or secondary MP, which form by being broken down into smaller sizes via weathering processes such as ultraviolet (UV) radiation exposure, biological activity and mechanical action.

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Beaches are emerging hotspots for MP pollution globally allowing for an enhanced understanding of both localised and distant sources and transportation vectors of MPs. Although this increasing plastic pollution in coastal environments threatens beach aesthetics, MPs are becoming understood to be more than simple inert particulates which often contain hazardous plastic additives (Henkel *et al.* 2023). In addition, MPs have been found to act as vectors for pollutant transport such as persistent organic pollutants, toxic metals and toxic microalgae (Casabianca *et al.* 2019). There is also concern about their consumption by marine organisms that mistake MPs as food sources due to their colouration and morphology or becoming passively consumed by benthic filter feeders (Duncan *et al.* 2019). MP consumption by fish have been shown to weaken abilities to avoid predation, damage gills and digestive tracts, create false senses of satiation, reduce reproductive fitness and ultimately, result in increased mortality (Yin *et al.* 2018). Adverse effects on human health from MPs and their associated additives are still being investigated. In beach environments, MP abundances have been observed to be increasing over time, as was observed by Ryan and Moloney (1990) across South Africa between 1984 and 1989. Consequently, establishing baselines for beach MP pollution has become increasingly necessary to enable a better understanding of these emerging contaminants.

Globally, beaches have displayed variable MP abundance (Browne *et al.* 2010; Liebezeit and Dubaish 2012; Dou *et al.* 2021; Laglbauer *et al.* 2014; Nel and Foreman 2015; Alvarez-Zeferino *et al.* 2020; Nhon *et al.* 2022 and Yu *et al.* 2018). On the island of Ireland, MPs have been assessed within lake environments (Nava *et al.* 2023) and Williams *et al.* (2017) investigated the nature and sources of beach litter but not MPs. In addition, MPs have been assessed in riverine (Murphy *et al.* 2022), atmospheric (Roblin *et al.* 2020) and offshore marine environments (Lusher *et al.* 2014). MP pollution was assessed in a variety of marine sediment types from the Irish coast (Marques Mendes *et al.* 2021) and found to be more concentrated in intertidal than subtidal environments.

The aims of this paper are to assess MP pollution on six Northern Ireland (NI) north coast beaches, to identify likely MP pollution sources and identifying the potential role of environmental and anthropogenic influences on MP morphologies, diversities, and relationships. These aims were met through the following objectives.

- Summer and winter sampling of six beaches with differing environmental settings and anthropogenic influences.
- Quantifying and classifying MP abundance, morphology and colouration through visual assessment, aided by fluorescence microscopy.
- Analysing spatial and temporal differences in MP abundances.
- Analysing MP abundance and wave energy relationships along beaches progressing eastward through regression analysis.

2 Study Area

NI's diverse coastline (Cooper 2011) is composed of alternating rocky headlands and embayments, resulting in the formation of a succession of discrete headland-bay beaches

(Jackson and Cooper 2010). Atlantic swell is refracted around the north coast, creating high-energy, swash-aligned beaches in the west. Furthermore, mean spring tidal range increases moving from west to east (Jackson *et al.* 2022). Additionally, waves become less energetic gradually moving eastward with declining significant wave heights occurring across Ireland's north coast (Jackson and Cooper 2010; Gallagher *et al.* 2014). Winter storms exhibit wave heights between 10-12m (Loureiro and Cooper 2019).

A variety of embayed beaches were assessed over a seasonal period, namely Portstewart Strand (PS), Portrush West (PR), Runkerry Beach (RB), Whitepark Bay (WB), Murlough Bay (MB) and Cushendun Beach (CB) (Figure 1). These beaches are subject to different anthropogenic pressures. In terms of population PS, is adjacent to nearby towns Portstewart and Castlerock with populations of 7854 and 1155 people, respectively. PR is near Portrush with 6150 people and RB is located beside Portballintrae with 500 people (NISRA 2023). These populations, in close proximity to the beach, resulted in these being designated as urban beaches. WB is located close to Ballintoy with a population of 150 people, MB is located near Ballyvoy with a population of 72 people and CB is located beside Cushendun with a population of 138 people (NISRA 2024). WB, MB and CB were therefore designated rural beaches.

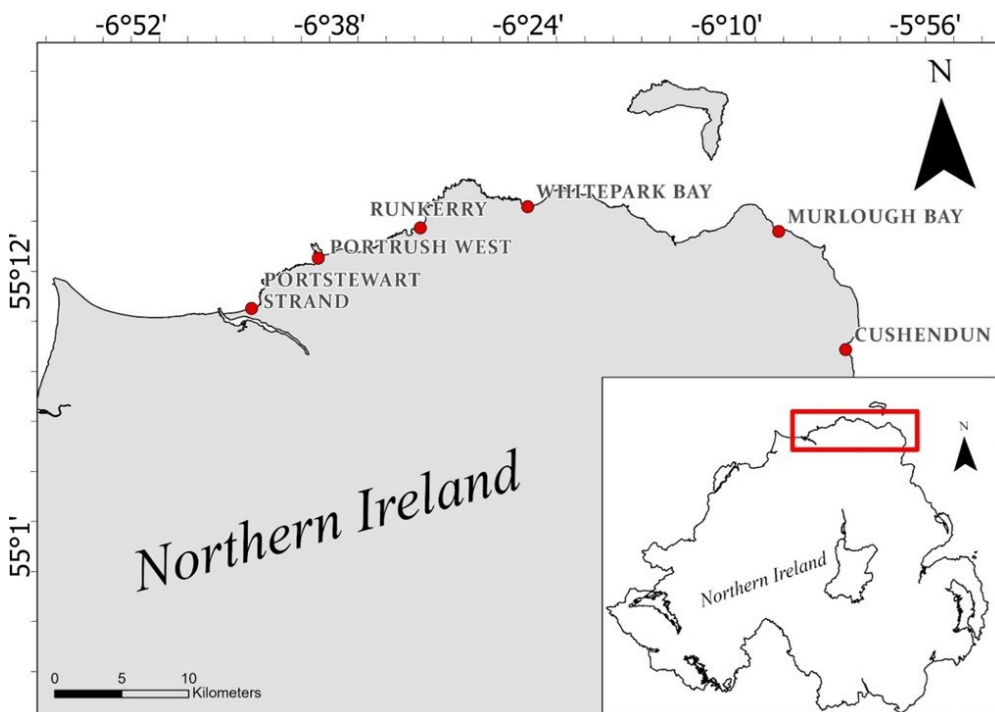


Fig. 1. Location of the six beaches sampled.

3 Methodology and Research Design

Beach sediment collection, MP extraction, MP identification and quantification and statistical analysis procedures are outlined below which enabled an understanding of MP abundances and distributions on NI beaches.

3.1 Sampling Procedure

Data was collected during August 2023 and January 2024 by sampling along a shore-parallel 100m transect at each beach's high tide line (HTL) (Rabari *et al.* 2022). Four samples were taken along the 25m, 50m, 75m and 100m points in the transect to reach a suitable confidence level for statistical analysis (Besley *et al.* 2017). This resulted in four sediment samples being collected per beach within each season. At each 25m interval along the transect, a 1m² quadrat was placed. Five subsamples were taken at maximum depth of 5cm in each quadrat corner and from its centre (Bissen and Chawchai 2020). Overall, a total of 1kg of wet sand was collected per 1m² quadrat. Due to this study's sampling of sediment mass within a defined 1m² square area the MP abundances were reported as MP/kg/m². This sampling of both sediment mass and square area allowed for suitable comparisons between studies which measured MPs by square area and sediment mass independently. Organic matter (OM) was avoided to aid later laboratory extraction and quantification when using Nile Red dye. GPS co-ordinates were recorded for seasonal sampling repetition.

3.3 Laboratory Extraction of MPs

Samples were dried at 60°C for 48 hours, passed through a 3.35mm sieve and 50g subsamples were prepared (Besley *et al.* 2017) (Figure 2). Density separation of MPs was performed using a mixed solution of five litres of deionised water and 358.9g of table salt (Rabari *et al.* 2022; Besley *et al.* 2017). Solutions were poured through a 60mm Whatman glass microfiber GF/A 1.6µm pore size filter paper to ensure table salt MPs larger than 1.6µm were trapped and discarded. A filtered solution (200ml) was poured into each sub-sample beaker, which were stirred vigorously for two minutes and then left to settle for two hours. The supernatant was decanted and filtered through a vacuum filter with 1.6µm filter papers. Beakers were additionally rinsed with deionised water to ensure trapped MPs on beaker glasses sides were fully captured. A solution (4ml) of 30% H₂O₂ was added to each filter paper to remove any OM (Prata *et al.* 2019). Digestion took place over seven days to ensure full OM removal. The beakers were vacuum filtered again and rinsed using deionised water to remove any remaining H₂O₂ from the MP. MPs on filter paper were placed on glass petri dishes for microscopic analysis.

3.4 MP Contamination Control

Procedural blank petri dishes with filter paper were exposed to the air in the laboratory throughout all laboratory work to assess possible airborne MP contaminants. Plastic

items were avoided during sampling and laboratory work with wooden, glass and metal equipment substituted where possible. Sub samples were covered with aluminium foil to prevent airborne MP contamination. Cotton clothing and blue nitrile gloves were worn throughout MP sampling, extracting and analysing.

3.5 Quantification and Characterisation of MP

MPs were investigated under a 40X magnification microscope. MPs were counted manually and were visually categorized by six colours (red, blue, black, white, green, clear) and five shapes (film, foam, pellet, fragment or fibre) to aid in the identification of likely MP sources as well as the range of potential sources through diversity analysis. Fibres could be linked to originating from textiles/fishing gear and fragments from tourism litter breakdown or other general plastic breakdown. Foams and films could originate from food packaging and pellets possibly from industrial plastic nurdles. It is worth highlighting that spectroscopic analysis is needed to gain a more robust understanding of likely sources. Furthermore, identifying MP morphologies allowed for the identification of transportation mechanisms. Nile Red dye was used to aid MP quantification and classification. The dye was diluted at 1mg/ml in methanol and stained filter papers were left to incubate for 30 minutes at 30°C in darkness (Shruti *et al.* 2022). The MPs were exposed to a UV torch emitting a 405nm wavelength light. Due to this study's sampling of both square area and sediment mass, all abundance results of MPs were presented as MP per kg/m² for comparison with other sites which used both kg and m² independently.

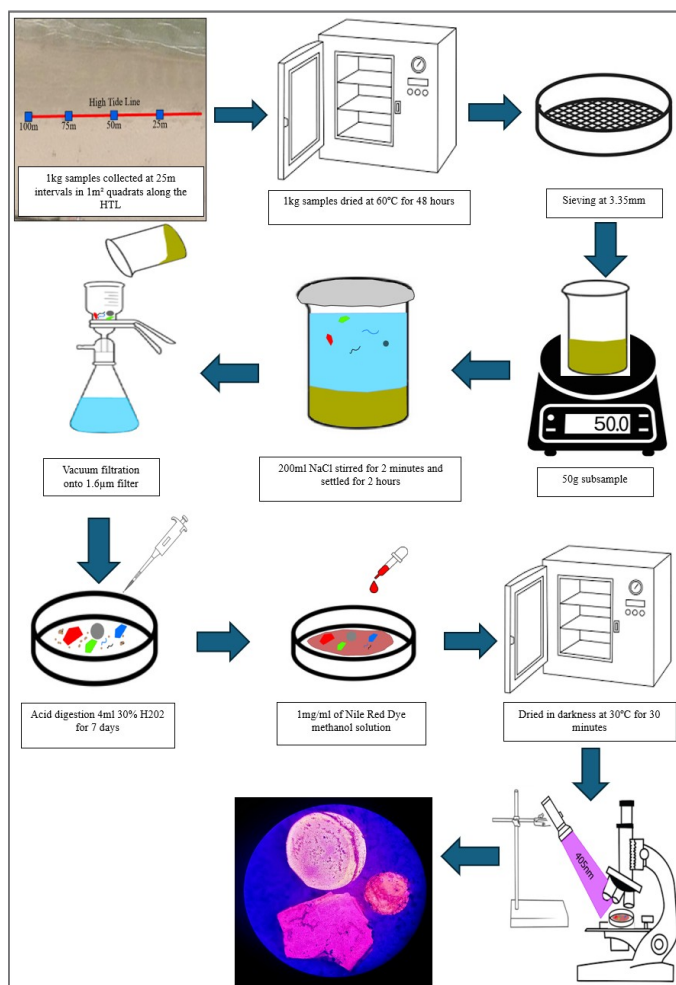
3.6 Statistical Analysis

A statistical test of normality was performed through a Shapiro-Wilk Test which confirmed the data followed a normal distribution. Subsequently, this was followed by a Two-Way ANOVA test of significance at an alpha value of 0.05 using IBM SPSS v28.0.1.1 (15). The Two-Way ANOVA test assessed differences in beach MP abundances regarding seasonality and rurality. Regression analysis was performed to assess whether a relationship existed between wave energy and MP abundances moving east. Distance moving eastward was used as a proxy to reflect declining wave energy. Distance was chosen as a viable wave energy proxy due to declining significant wave heights moving eastward along Ireland's north coast (Jackson and Cooper 2010; Gallagher *et al.* 2014). It is worth noting this method does have limitations in that it omits other wave energy altering factors like local wind conditions and storm events.

R Studio 3.3.0 was used to create an RDA plot of MP morphologies in relation to multiple environmental variables. To create an RDA plot, rivers were classified utilising presence and absence values. Hydrometric station data was not available across all study area rivers or other sources of river discharge values. To overcome this, river width was used as a proxy for river discharge to ensure all rivers were not treated as equally impactful. However, it is worth noting limitations to this approach as river width does not factor tidal processes or river depth/velocity. River mouth width measurements were obtained

via satellite imagery measurements within ArcGIS Pro 2.7.0. Embayment beach presence was noted, and storm water outlet presence and abundance were included, based on data from NI Water (2024). Urbanisation and rurality magnitude was based on nearest town populations. ArcGIS Pro 2.7.0 was used to display MP abundances, morphologies, and colourations across the sites to aid in highlighting spatial and temporal differences. MP polymer diversity among each beach was analysed using the Shannon-Wiener Diversity Index (*SDI*) to highlight beaches with potentially higher or more varied pollution sources and transportation vectors. Polymer diversity values were calculated by considering each colour and morphology of the MPs and their relative abundances at the individual beach level, urban/rural level and NI level and across summer, winter and annual timescales. Each combination of morphology and colour was treated as a unique category in the diversity calculation.

$$SDI = - \sum_{i=1}^S p_i \ln(p_i)$$



Where *SDI* is the diversity of MPs per beach, *S* is the sum of all unique MP morphology and colouration combinations per beach, *i*=1 denotes the first group of MP morphology and colourations observed and is the proportion of each MP morphology and colouration group's total abundance relative to the total abundance of MPs per beach.

Fig. 2. Diagram of methodologies for the collection, extraction and quantification of MPs in beach sediments.

Results

4.1 Microplastic Abundances

A total range of 0-380 MPs per kg/m² was observed across all beaches annually in NI (0-380 for urban beaches and 0-240 for rural beaches, Table 1). Two outlier values of 380 MPs per kg/m² in urban beaches in the summer and 240 MPs per kg/m² in rural beaches in the winter were observed (Table 1, Figure 3). PS possessed the highest mean MP abundance in the summer ($m=220$) and CB the highest mean in the winter ($m=160$). Urban beaches possessed their highest annual mean abundance during the summer ($m=187$) and possessed a higher mean abundance than rural beaches which had their annual highest mean abundance in the winter (winter $m=100$), (Figure 3).

Urban beaches declined in mean abundance during the winter ($m=42$) being lower in average abundance than rural beaches during the winter (Figure 3). A noticeable easterly decline in MP abundances was present in summer but absent in winter (Figure 4). Two-Way ANOVA results revealed a significant relationship between MP abundance and beach rurality ($p=0.004$, Table 2). Significantly higher abundances were present within urban beaches. There was a significant difference in abundance and seasonality ($p=0.03$, Table 2). Significantly higher abundances were present during the summer.

Table 1. Beach characteristics and respective MP abundances across summer and winter with maximum/ minimum and mean MP (kg/m²).

Beach Name	Beach Nature	Tidal Range	Typology	Summer MP/kg/m ² (Min/Max)	Winter MP/kg/m ² (Min/Max)	Summer MP/kg/m ²	Winter MP/kg/m ²
Portstewart	Urban	Micro-tidal	Embayment	100–380	0–100	220	70
Portrush West	Urban	Micro-tidal	Embayment	140–220	0–60	185	15
Runkerry	Urban	Micro-tidal	Embayment	100–280	20–60	155	40
Whitepark Bay	Rural	Micro-tidal	Embayment	0–60	40–60	35	75
Murlough Bay	Rural	Micro-tidal	Embayment	0–40	0–120	20	65
Cushendun	Rural	Micro-tidal	Embayment	20–60	100–240	35	160

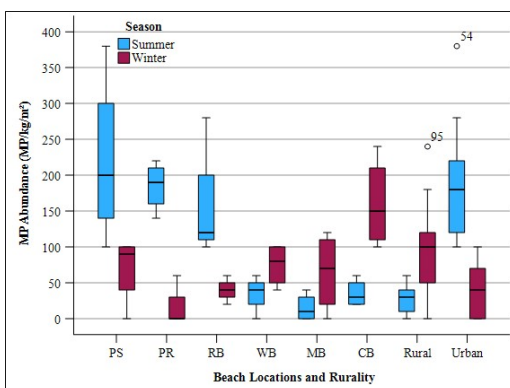


Fig. 3. Boxplot of MP abundances (MP per kg/m²) in relation to each beach, rurality and seasonality. Boxes represent the 25th and 75th quartiles and whiskers represent the 10th and 90th percentiles. Two outliers were observed across winter rural beaches and summer urban beaches. Solid lines within each box indicate median abundances observed.

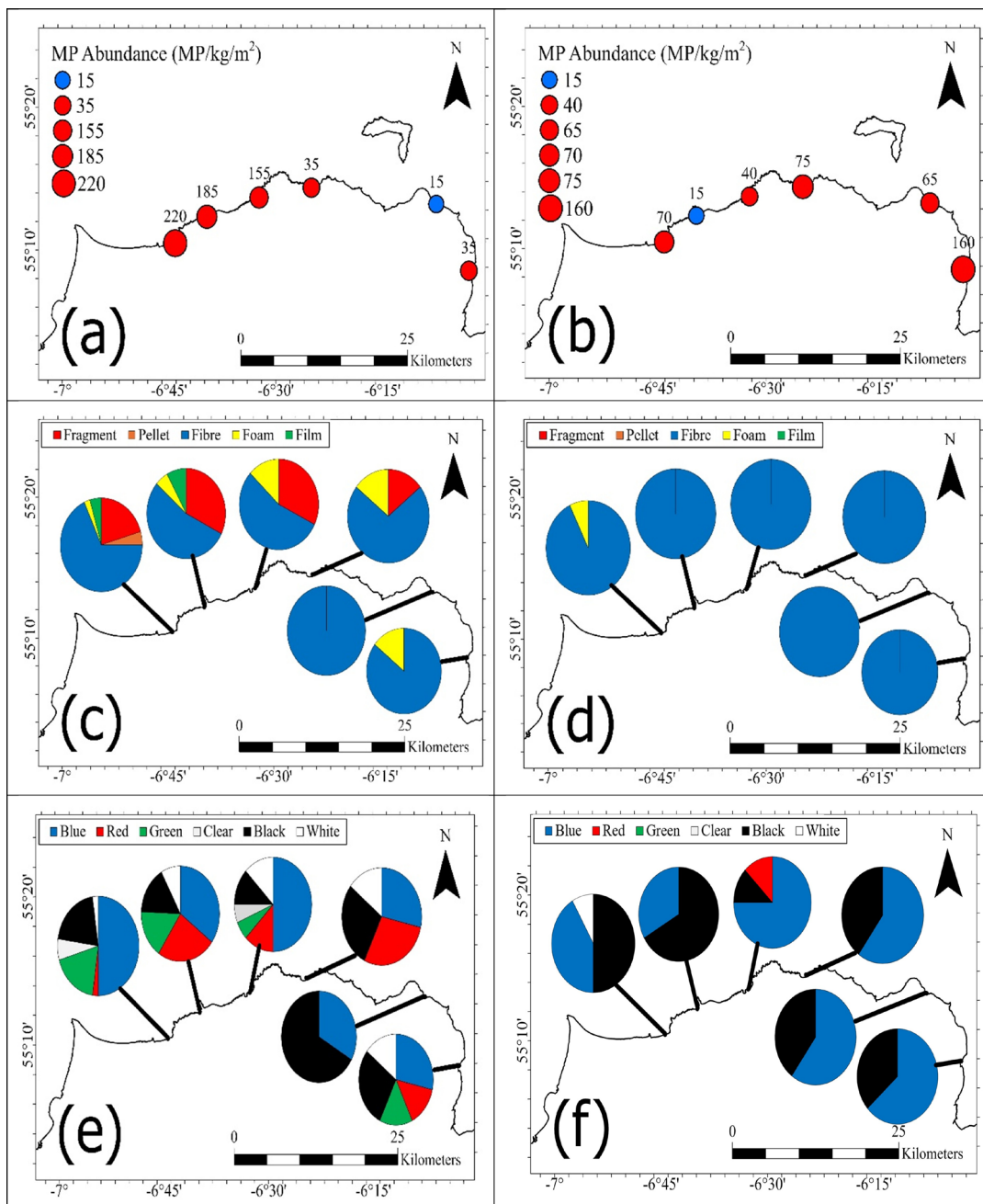


Fig. 4. Maps displaying the average abundance of MP, percentage of MP morphologies observed and MP colourations per beach in the summer (left panels) and winter (right panels), in MP per kg and m². (a) summer MP abundance. (b) winter MP abundance on each beach. (c) summer MP morphologies. (d) winter morphologies on each beach. (e) summer MP colourations. (f) winter colourations on each beach.

Table 2. A Two-Way ANOVA assessment of the significance of MP abundance in relation to beach rurality and season. The alpha value was set to 0.05.

Source	df	F	P
Corrected Model	3	19.338	<.001
Intercept	1	118.419	<.001
Rurality	1	9.309	.004
Season	1	5.006	.030
Error	44		
Total	48		

Dependent Variable: MP Abundance;
R Squared = .569 (Adjusted R Squared = .539); *df* = degrees of freedom;
F = Statistic; *P* = Significance values <0.05.

In terms of MP abundance NI (0-380 MPs/kg/m²) had similar levels of MP pollution to the USA (43-443 MPs/kg, Yu *et al.* 2018) and the Republic of Ireland (0-553 MPs/kg, Marques Mendes *et al.* 2021) (Table 3). NI had higher MP abundances in comparison to Vietnam (0-26.32 MPs/m², Nhon *et al.* 2022) and lower abundances than beaches in China (67-27,000 MPs/kg, Dou *et al.* 2021). However, variations in MP abundance across sites may reflect differences in study methodologies rather than actual beach pollution.

Table 3. MP abundance ranges on beaches globally measured in MP or kg per m².

Country	Location	Particle size	MP Abundance	Unit	Reference
Portugal	Beach	1.2 µm–5mm	1-137	m ²	Browne <i>et al.</i> 2010
Germany	Beach	1.2 µm–5mm	671	kg	Liebezeit and Dubaish. 2012
China	Beach	0.25-5mm	67-27,000	kg	Dou <i>et al.</i> 2021
Slovenia	Beach	0.25-5mm	177.8	kg	Laglbauer <i>et al.</i> 2014
South Africa	Beach	0.065-5mm	688.9–3308	m ²	Nel and Foreman. 2015
Mexico	Beach	1-5mm	31.7–545.8	m ²	Alvarez-Zeferino <i>et al.</i> 2020
Vietnam	Beach	0.5mm-5mm	0-26.32	m ²	Nhon <i>et al.</i> 2022
United States	Beach	1µm–5mm	43–443	kg	Yu <i>et al.</i> 2018
Republic of Ireland	Beach	63 µm–5mm	0–553	kg	Mendes <i>et al.</i> 2021
Northern Ireland	Beach	1.6 µm–3.35 mm	0–380	kg/m ²	This Study

4.2 Microplastic Morphology

Across both seasons MPs of all morphologies were observed (Figure 5). Fibres were the dominant morphology (76.6%; $n=164$) (Figure 4, 6), followed by fragments (15% ; $n=32$), foams (5.1%; $n=11$), films (2.3%; $n=5$) and pellets (0.9 %; $n=2$). During summer, fibres were again dominant (62.8% $n=81$), followed by fragments (24.8%; $n=32$), foams (7%; $n=9$), films (3.9%; $n=5$) and pellets (1.6%; $n=2$). In winter, fibres remained dominant (97.6% ; $n=83$) followed by foams (2.4%; $n=2$). No other morphologies were observed.

In urban beaches annually, fibres were dominant (66.4%; $n=91$) followed by fragments (22.6%; $n=31$), foams (5.8%; $n=8$), films (3.6%; $n=5$) and pellets (1.5%; $n=2$). In rural beaches annually, fibres were again dominant (94.8%; $n=73$), followed by foams (3.9%; $n=3$), and fragments (1.3%; $n=1$) with no films or pellets being observed.

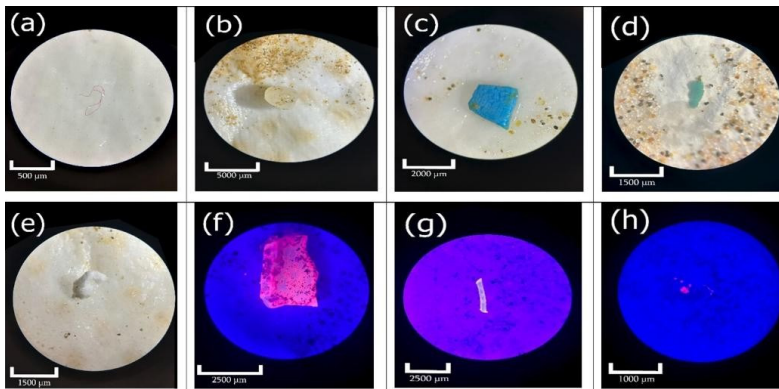


Fig. 5. Representative images of each MP category identified. The following categories were found (a) fibre (b) pellet (c) fragment (d) film (e) foam. Panels f-h show fluorescence microscopy imagery and illustrate its ability to enhance MP identification and quantification.

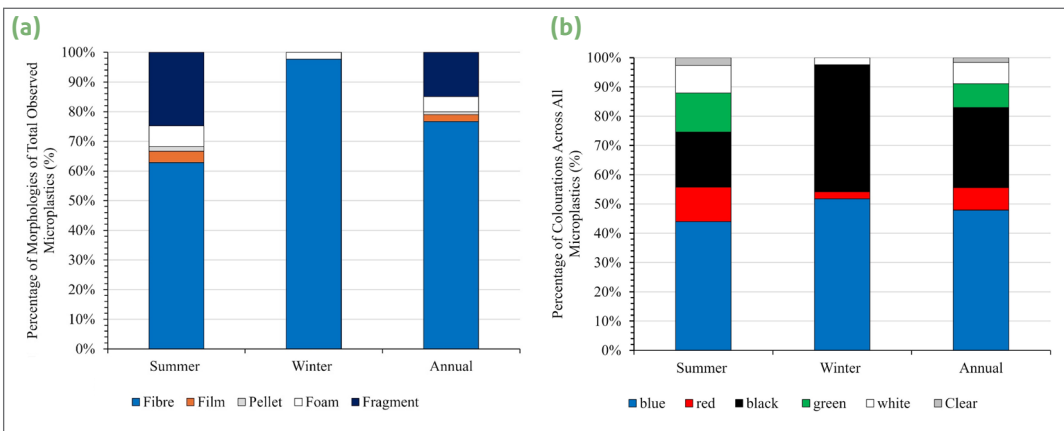


Fig. 6. (a) Percentage of MP morphologies observed across both summer and winter across all beach sites in NI and their associated predominant colourations. (b) The percentages of MP colours observed in both summer and winter across all beach sites in NI. Blue MPs were most common annually.

4.3 Microplastic Colouration

Blue was the most observed polymer colour across both seasons (46.7%, Figure 6; Table 4), and blue fibres was the most widely observed colouration and morphology combination (40.7%), followed by black fibres (28.5%) and blue fragments (4.7%). In summer blue was the most observed polymer colour (43.4%), followed by black (18.6%) and green (13.2%). Blue fibres were the most widely observed colouration and morphology combination (33.3%) of all summer MPs, followed by black fibres (18.6%) and blue fragments (7.8%). During the winter, blue was the most observed polymer colour (51.8%) followed by black (43.5%) and joint third being white and red MPs (2.4%). Blue fibres were the most widely observed colouration and morphology combination of all winter MPs (51.8%), followed by black fibres (43.5%) and then white foams and red fibres joint third (2.4%).

Within urban beaches annually, blue was the most observed polymer colour (45.3%), followed by black (21.9%) and green (11.7%). Blue fibres were the most widely observed colouration and morphology combination of all urban MPs (35.8%), followed by black fibres (21.9%) and then blue fragments (7.3%). Within rural beaches annually, blue was the most observed polymer colour (49.4%), followed by black (40.3%) and red at (5.2%). Blue fibres were the most widely observed colouration and morphology combination of all rural MPs (49.4%), followed by black fibres (40.3%) and joint third were red fibres and white foams (3.9%).

Table 4. Total MPs morphologies and colourations observed in each beach across summer and winter and their potential sources. The order of numbering denotes colourations observed (e.g. *Bl*: Blue; *Bk*: Black; *Rd*: Red; *Gr*: Green; *Wh*: White; *Cl*: Clear)

Beach	Season	Fibres (Bl,Bk, Rd,Gr)	Fragments (Bl,Rd,Gr, Wh,Cl)	Foams (Wh,Bl)	Films (Bl,Wh, Gr)	Pellets (Cl)	MP Potential Sources
Portstewart	Summer	18,9,0,3	2,1,5,0,1	0,1	1,1,0	2	Fibres: Textiles/Fishing Gear; Fragments: Tourism Litter Breakdown/Other Plastic Breakdown; Foams and Films: Food Packaging; Pellets: Industrial Plastic Nurdles
Portstewart	Winter	6,7,0,0	0,0,0,0,0	1,0	0,0,0	0	Fibres: Textiles/Fishing Gear; Foams: Food Packaging
Portrush	Summer	8,6,3,3	4,5,2,1,0	2,0	1,1,1	0	Fibres: Textiles/Fishing Gear; Fragments: Tourism Litter Breakdown/Other Plastic Breakdown; Foams and Films: Food Packaging
Portrush	Winter	1,2,0,0	0,0,0,0,0	0,0	0,0,0	0	Fibres: Textiles/Fishing Gear
Runkerry	Summer	12,3,1,1	4,2,1,1,2	4,0	0,0,0	0	Fibres: Textiles/Fishing Gear; Fragments: Tourism Litter Breakdown/Other Plastic Breakdown; Foams: Food Packaging
Runkerry	Winter	6,1,1,0	0,0,0,0,0	0,0	0,0,0	0	Fibres: Textiles/Fishing Gear
Whitepark Bay	Summer	2,2,1,0	0,1,0,0,0	1,0	0,0,0	0	Fibres: Textiles/Fishing Gear; Fragments: Tourism Litter Breakdown/Other Plastic Breakdown; Foams: Food Packaging
Whitepark Bay	Winter	9,6,0,0	0,0,0,0,0	0,0	0,0,0	0	Fibres: Textiles/Fishing Gear
Murlough Bay	Summer	1,2,0,0	0,0,0,0,0	0,0	0,0,0	0	Fibres: Textiles/Fishing Gear
Murlough Bay	Winter	8,5,0,0	0,0,0,0,0	0,0	0,0,0	0	Fibres: Textiles/Fishing Gear
Cushendun	Summer	2,2,1,1	0,0,0,0,0	1,0	0,0,0	0	Fibres: Textiles/Fishing Gear; Foams: Food Packaging
Cushendun	Winter	19,12,0,0	0,0,0,0,0	1,0	0,0,0	0	Fibres: Textiles/Fishing Gear; Foams: Food Packaging

4.5 Microplastic Diversity

PR exhibited the highest summer diversity in polymer morphology and colouration ($SDI=2.25$, Table 5) and the lowest polymer diversity was at MB ($SDI=0.64$). In winter PS had the most diverse MP assemblage ($SDI=0.71$), and MB had the least ($SDI=0.27$). Urban beaches had higher diversity overall ($SDI=2.11$), in comparison to rural beaches ($SDI=1.34$). During the winter, urban beaches again showed higher diversity ($SDI=0.97$) than rural beaches ($SDI=0.62$). MP diversity was higher in the summer ($SDI=2.09$) than during the winter ($SDI=0.86$).

Table 5. MP polymer richness values attained through the Shannon-Wiener Diversity Index (SDI) assessing the abundances of different morphology and colouration combinations observed for each beach, urban and rural beaches collectively as well as NI's MP diversity collectively.

Polymer Diversity	Summer (SDI)	Winter (SDI)	Annual (SDI)
Portstewart	1.83	0.71	1.71
Portrush West	2.25	0.60	2.40
Runkerry	1.71	0.64	1.83
Whitepark Bay	1.55	0.66	1.26
Murlough Bay	0.64	0.27	0.69
Cushendun	1.27	0.63	1.05
Urban beaches	2.11	0.97	2.01
Rural beaches	1.34	0.62	1.13
All Beaches	2.09	0.86	1.88

4.6 Microplastic Abundance and Morphology Driving Factors

Polynomial regression analysis revealed a moderate negative correlation ($r^2=0.6596$, Figure 7) between summer MP abundance and wave energy, although this relationship was not strictly linear with a notable decline in MP abundance after RB with MPs rarely exceeding 50 MP/kg/m² in eastern beaches. Winter showed a similar relationship but with a moderate positive correlation ($r^2=0.4459$) and an increase in MP/kg/m² after RB again indicating MP abundances are not linearly correlated with wave energy.

RDA analysis revealed that in the summer, fragments were strongly positively related to increasing urbanisation with foams and films also sharing a weak positive relationship with urbanised beaches (Figure 8). Fragments, foams, and films were also negatively related to rural beaches. No relation between rurality, embayment presence and polymer morphologies was observed. Fibres displayed a strong positive relationship between river presence and increasing abundance in storm discharge points. During the winter, foams weakly positively related to urbanised areas and fibres displayed a weak relationship with river presence, embayment presence and rurality.

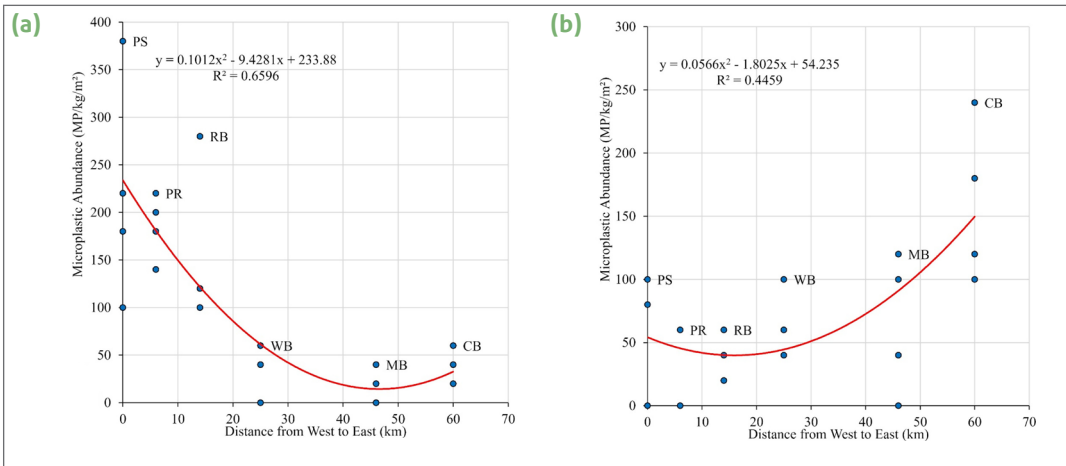


Fig. 7. Polynomial regression graphs displaying MP abundance in relation to decreasing wave energy associated with increasing distance east. (a) Summer MP abundance in relation to decreasing wave energy. (b) Winter MP abundance in relation to decreasing wave energy.

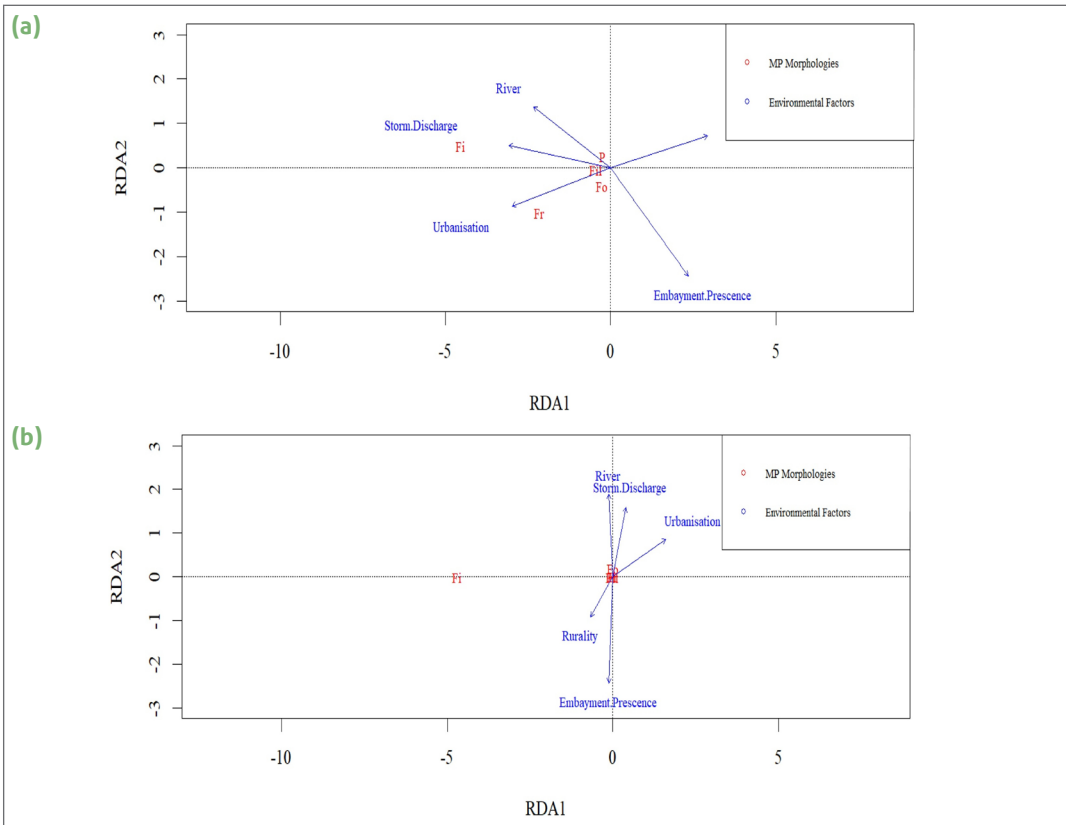


Fig. 8. RDA plots of MP morphologies across both (a) summer and (b) winter in relation to possible influencing variables regarding polymer abundance. Polymer labelling is according to polymer type (e.g. Fi: Fibre; Fo: Foam; Fil: Film; Fr: Fragment and P: Pellet).

5 Discussion

Urban and Rural Beach Differences

The current study reveals significantly higher abundance of MPs in urban beaches ($p=0.004$, Table 2 & Figure 3). This matches findings elsewhere (Zhang *et al.* 2024; Marques Mendes *et al.* 2021) that MP abundance increases with proximity to urban areas. The rural beaches in this study are less easily accessible and experience lower recreational pressure (Botero *et al.* 2013). In contrast, the urban beaches are close to highly populated areas with high numbers of storm water outlets. This proximity facilitates beach accessibility in conjunction with an abundance of amenities. Despite urban beaches having significantly higher MP abundances annually, they declined in mean abundance in the winter whereas rural beaches had a higher mean abundance in winter compared to summer. The reason for an expected urban decrease likely relates to lower winter tourism use, however, the reason for an increase in rural beaches is unclear but may be related to increased wave energy.

The higher abundance of MP on urban beaches may also be linked to the presence of rivers which behave as major MP pollution transportation vectors (Lloyd-Jones *et al.* 2023). For example, PS as an urban beach, had the highest mean abundance ($m=220$), which may be linked to the mouth of the River Bann. The Lower Bann is Lough Neagh's only river outlet, and Lough Neagh ranked the 11th most MP polluted lake of 38 sites globally (Nava *et al.* 2023).

The unexpected observation of the highest mean winter abundance ($m=160$) on a rural beach (CB) may also be due to the presence of a river. All the urban and rural beaches studied were embayments, suggesting that the observed differences were likely not due to embayment presence alone. However, embayment typology cannot be completely disregarded as a contributing factor to differing MP abundances, given that geomorphic characteristics such as size, orientation, and slope may vary significantly across beaches, meaning their impacts cannot be considered uniform. Consequently, despite these potential differences in embayment configurations, geomorphology alone is unlikely to fully explain the observed differences in MP abundance. Instead, MP abundance differences across beaches are more likely driven by each beach's unique localized pollution transportation vectors, such as the presence of rivers, large populations, and stormwater outlets. Given the dominance of observed fibres, it is also possible a large proportion of MPs may have originated from marine fishing gear, such as nets/ropes as was found by Zhang *et al.* (2021). Furthermore, offshore current patterns across each beach may vary. Consequently, observed beach MP abundances differences may also be influenced by dynamic offshore current vectors which may not have acted in a strictly linear manner moving eastward resulting in a more complex distribution of MPs. However, given that wave energy shared moderate correlations with MP abundances, wave energy also appeared to act as a driver of variations in MP abundances across beaches.

Seasonal Differences

The results show a significant difference between summer and winter abundances ($p=0.03$, Table 2 & Figure 3). This difference is consistent with the findings of Zhang *et al.* (2022) and Korez *et al.* (2019). However, other authors (Jaouani *et al.* 2022 and Chen and Chen. 2020) found no seasonal differences and Ariefdien *et al.* (2024) recorded higher abundances in winter. For NI, the current study's findings contrast with those of Williams *et al.* (2017), who observed no seasonal difference in general beach litter abundance. This could be explained by annual clean-up efforts targeting larger litter items whereas MPs are unaffected due to their size.

Significantly higher abundance of MPs in summer supports the possibility of seasonal tourism acting as a source as noted elsewhere by Bentaallah *et al.* (2024) and Quesadas-Rojas *et al.* (2021). There is a large influx of summer tourists to coastal towns of Portstewart and Portrush. This might promote an increase of macro plastics that later degrade into MPs or via direct MP inputs (Luan and Wang 2024; Zhang *et al.* 2022). Significantly higher summer MP abundances could also be explained by the first flush phenomenon (Chouchene *et al.* 2021). NI experienced 185.4mm of rainfall during July 2023, being twice the average monthly rainfall making it the wettest July on record (Met Office 2023). This seasonally abnormal rainfall likely resulted in an atypical amount of surface runoff, particularly in urban sites, thus promoting a sudden surge in high MP transportation to beaches. The dominance of fibres observed in this study could possibly be due to fibrous MPs being more easily transported by runoff as observed by Chouchene *et al.* (2021).

The lower winter abundance could be linked to the occurrence of storms Isha and Jocelyn during January 2024 (Met Office 2024), which may have enhanced aeolian transport and wave erosion and resulted in MP loss from beaches as noted elsewhere by Expósito *et al.* (2021). In contrast, Harley *et al.* (2022) observed increased post-storm MP abundances. The presence of seawalls in Portrush has been observed to promote offshore transport of sediments (Jackson and Cooper, 2010). This sediment transport could also include MPs, particularly when highly energetic storm waves reflect off the seawalls. The interaction between the seawalls and storm waves could have resulted in low energy dissipation and wave redirection seaward, promoting offshore transport of sediments and possibly MPs. Furthermore, it is also possible winter storms may have simply redistributed MPs that are normally concentrated on beach HTLs. Given this study's focus on the HTL, winter MPs may have been redistributed among different beach face zones.

Regression analysis shows moderate relationships during summer and winter between wave energy and MP abundance (summer $r^2 = 0.6596$, winter $r^2 = 0.4459$, Figure 7). The moderate seasonal relationships suggest that spatial variation in wave energy has a moderate impact on MP abundance. Other authors such as Bissen and Chawchai (2020) have noted an increase in MP abundances in relation to declining wave energy. The embayed nature of the beaches in the study area may limit such effects. The moderate negative summer relationship indicates that wave energy does have some impact on abundance, but this effect may also act alongside the general decline in population toward the east.

Microplastic Characteristics and Origins

Fibrous MPs were the most common polymer morphology (76.6% of all MPs annually) within beaches (Figure 6). This aligns with the findings of Azaouaj *et al.* (2024) and Luan and Wang (2024) but contrasts with the findings of Karthik *et al.* (2018) and De-la-Torre *et al.* (2020) who observed other dominant morphologies. The use of acid digestion and Nile Red dye to eliminate organic fibres in this study means that plastic fibres can be deemed the dominant MP morphology on NI's beaches. The dominance of blue fibres is consistent with assessments across Ireland where blue fibres were the most common MP form in rivers (Murphy *et al.* 2022) and offshore waters (Lusher *et al.* 2014). This study's blue fibre dominance also aligns with a continental assessment of MPs across beaches in Europe (Lots *et al.* 2017). It contrasts, however, with the findings of Marques Mendes *et al.* (2021), who observed colourless fibres as most common across Irish beaches.

RDA analysis showed that fibres were positively related with river presence and storm water outlets. Since the majority of NI's MP pollution was fibrous, this suggests that much of it originates from these transportation vectors (Figure 8). Ariefdien *et al.* (2024) and Lloyd-Jones *et al.* (2023) also noted rivers and storm water outlets acted as major beach MP pollution transportation vectors. In the study area, rivers, urban areas and storm water outlets often coincided with one another. Consequently, further research is required to investigate these transportation relationships to determine individual transportation vector contributions to beach MP pollution. It is hypothesised that the dominance of blue fibre colourations could originate from clothing materials, due to the common industrial application of blue dye within textiles (Ronda *et al.* 2023). Equally, blue fibres may also have originated from marine fishing gear due to blue fishing line having been utilised for fishing purposes (Zhang *et al.* 2021). However, further research is needed for more robust MP source confirmation.

A positive relationship between beach urbanisation and fragment MPs in the summer also existed (Figure 8). This finding, in conjunction with significantly higher MP abundances in the summer, as well as the high abundance of fragments during the summer (24.8%) in comparison to winter (0%) aligns with De Falco *et al.* (2019)'s observations that summer tourist influxes are responsible for fragmented MP prevalence on beaches. During the summer, blue was the most observed fragment colouration, a finding supported by Marques Mendes *et al.* (2021) and likely due to common dye applications.

Implications

These results are the first quantitative assessment of MP pollution on beaches in NI. Beach rurality ($p=0.004$) and seasonality ($p=0.03$) were significant influences on MP abundance. Seasonal tourism activities, storm water outlets and rivers all act as likely major transportation vectors of MP pollution across beaches. MPs were ubiquitous across all beaches in NI throughout both the summer and winter.

The finding that NI's beach MPs were dominated by fibres supports the work of others that identifies fibres as the most common MP morphology on beaches globally and raises concerns regarding the potential impacts on marine fauna. Fibrous MPs can significantly inhibit juvenile *organisms* from consuming food sources (Cole *et al.* 2019) and pose a risk to marine organisms. Moreover, blue MPs (the most common form in this study) are more likely to be consumed by marine fauna than other colourations (Duncan *et al.* 2019).

The higher MP diversity observed across urban beaches annually ($SDI= 2.01$) compared to rural beaches ($SDI= 1.13$) and higher MP diversity during the summer ($SDI= 2.09$) compared to the winter ($SDI= 0.86$) act as indicators of a larger range of potential MP sources and transportation vectors across differing locations and timeframes (Bimali Koongolla *et al.* 2018) (Table 5). As a result, urban beaches during the summer likely possess the largest range of pollution sources and will be the most potentially harmful but difficult to manage for tackling plastic pollution.

Limitations and Future Recommendations

While this study successfully identified likely MP pollution-driving factors, it was not able to quantify individual contributions to beach MP pollution. To expand on this, future research is needed to quantify MP inputs from transportation vectors like river and storm water outlets. This study focussed on the HTL and knowledge of variable MP abundances across differing shore zones remains limited. Future studies are needed to consider the subtidal, intertidal and supralittoral zones of beaches. Furthermore, whilst this study could identify possible MP sources based upon morphologies and could also identify the range of MP sources based on diversity analysis. This study could only speculate regarding MP source materials. To address this source knowledge gap, future studies should utilise spectroscopic techniques to determine MP chemical origins to gain a more confident understanding of MP sources.

Regarding MPs and their relationships to wave energy, this study only assesses NI's high energy microtidal environment. and could be expanded to consider a wider range of beach types. Rural beaches provide a baseline to better infer how natural processes are independently responsible for MP deposition on beaches in the absence of anthropogenic influences. The study was limited to two seasons and lacks a longitudinal component. It does, however, offer a baseline against which future studies might be compared.

Finally, comparative studies of MPs in beaches are hampered by a diversity of methodologies in different studies and a standardized sampling methodology is required. Utilising smaller pore size filter paper and a 5mm mesh sized sieve would enable of the complete MP size range.

6 Conclusion

MPs were ubiquitous across the studied beaches. This study revealed significantly higher MP abundances on urban beaches than on rural beaches and higher abundances in summer. Urban beaches exhibited a greater diversity of MP colouration and morphology combinations annually in contrast to rural beaches, reflecting a wider range of pollution sources and transportation vectors. In addition, abundances displayed a relationship with wave energy; high MP values were found on western beaches in summer but the relationship with wave energy was non-linear. Analysis of MP morphology identified fibres are most common and their abundance is correlated with river and storm water outlet presence. Due to fragments being positively related to increased beach urbanisation and their declining winter abundance, summer tourism was identified as a likely pollution source. Regarding MP morphology, all morphologies were observed across NI's beaches, with fibres being the dominant polymer and blue being the most observed colouration. Blue fibres were the most common morphology/colouration combination. NI's beach MP pollution can be linked to multiple factors, namely beach proximity to largely populated urban settlements, seasonal tourism influxes and the presence of rivers and storm water outlets. MP pollution in NI beaches is an area of concern that should be considered by policy makers concerned with conservation of marine ecosystems and protection of human welfare.

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