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Long term trends in floating plastic pollution within a marine protected area identifies threats for Endangered northern bottlenose whales

Noreen E. Kelly[a,*], Laura Feyrer[a,b], Heidi Gavel[a,c], Olga Trela[a], Wayne Ledwell[d], Heather Breeze[a], Emmaline C. Marotte[a], Leah McConney[a], Hal Whitehead[b]

[a] Fisheries and Oceans Canada, Bedford Institute of Oceanography, Dartmouth, Nova Scotia, Canada
[b] Dalhousie University, Halifax, Nova Scotia, Canada
[c] St. Mary's University, Halifax, Nova Scotia, Canada
[d] Newfoundland and Labrador Whale Release and Strandings, Newfoundland, Canada

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ABSTRACT

"The Gully", situated off Nova Scotia, Canada, is the largest submarine canyon in the western North Atlantic. This unique oceanographic feature, which became a Marine Protected Area (MPA) in 2004, is rich in marine biodiversity and is part of the critical habitat of Endangered northern bottlenose whales (*Hyperoodon ampullatus*). To understand the potential impact of plastic pollution in the MPA and on this Endangered cetacean, we evaluated trends over time in the abundance and composition of plastics and compared these to the stomach contents of recently stranded northern bottlenose whales. From the 1990s–2010s, the median abundance of micro-sized (<5 mm) and small plastics (5 mm–2.5 cm) increased significantly, while the median abundance of large plastics (>2.5 cm) decreased significantly. Plastic abundance from the 2010s for micro-sized and small plastics varied from 5586–438 196 particles km^{-2}, higher than previously measured estimates for surrounding offshore areas. Polymers identified using FTIR spectroscopy included polyethylene, polypropylene, polyethylene terephthalate polyester, nylon, alkyds (paint), and natural and semi-synthetic cellulosic fibers. The abundance of large debris ranged from 0 to 108.6 items km^{-2} and consisted of plastic sheets and bags, food wrappers and containers, rope, fishing buoys, and small plastic fragments. Whale stomach contents contained fragments of fishing nets, ropes, bottle caps, cups, food wrappers, smaller plastic fragments, fibers, and paint flakes, consistent with the composition and character of items collected from their critical habitat. Despite being far from centres of human population, the unique oceanographic features of The Gully (i.e. currents and bathymetric complexity) may concentrate plastic debris, increasing exposure rates of whales to plastic pollution. The increase in micro-sized and small plastics over time suggests associated health and welfare impacts of ingested plastics should be accounted for in future recovery plans for this Endangered species.

1. Introduction

The first reports of noticeable quantities of floating plastic debris in the ocean occurred in the 1970s (Carpenter and Smith, 1972; Colton et al., 1974; Venrick et al., 1973). Since that time, a rapid, non-linear increase in total plastic production and use of plastic has occurred worldwide (Andrady, 2017; Geyer et al., 2017). Entering the ocean from multiple sources (Jambeck et al., 2015; Lebreton et al., 2022), plastic debris is now considered a ubiquitous marine pollutant, found in all oceans and throughout the water column, with the highest concentrations found floating at the surface or accumulating in sediments (Cózar et al., 2014; Egger et al., 2022; Van Cauwenberghe et al., 2015). The prevalence and distribution of plastic debris gives rise to concerns about its impacts on the marine environment and biota at all trophic levels (Galloway et al., 2017; Kühn and van Franeker, 2020). However, despite the rapid increase in studies of plastic pollution over the last decade (Schmid et al., 2021), major gaps in understanding of baseline plastic concentrations remain, particularly in the western North Atlantic (Haarr et al., 2022).

Plastic debris is defined by its size, colour, morphology, abundance, and polymer composition (Hartmann et al., 2019), features which indicate its source, fate, and impact (Andrady, 2017; Wright et al.,

* Corresponding author.
E-mail address: noreen.kelly@dfo-mpo.gc.ca (N.E. Kelly).

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2013). For marine mammals, particle size largely determines the degree and type of impact. Lethal impacts of large plastic debris (>2.5 cm) may result from entanglement or ingestion (e.g., blockage, suffocation, and starvation), while sublethal health impacts include injuries, infection, compromised feeding or digestive capacity, or malnutrition, all which may have long term fitness impacts on reproduction or growth (Baulch and Perry, 2014; Fossi et al., 2018; Senko et al., 2020; Zantis et al., 2021). Small (5 mm–2.5 cm) and micro-sized (1 μm–5 mm) plastic debris ingestion has also become a growing concern due to potential health impacts and animal welfare concerns (Eisfeld-Pierantonio et al., 2022), such as the transfer of chemicals and toxic additives from ingested particles into body tissues, further amplifying pollutant burdens (Hermabessiere et al., 2017; Ribeiro et al., 2019; Teuten et al., 2009). While the impacts of large debris (e.g., entanglement) have been more frequently studied across marine mammal species, the impacts of smaller and micro-sized plastics remain poorly understood (Fossi et al., 2018). In addition, the impact of plastic debris on cetaceans is relatively understudied compared to other marine mammal taxa (Zantis et al., 2021). Due to the difficulty in studying wild cetaceans living in remote offshore environments, the impacts of plastic debris have largely been restricted to opportunistic sampling of stranded whales when and where they occur.

In the western North Atlantic, a small, Endangered population of northern bottlenose whale (NBW, *Hyperoodon ampullatus*) inhabits offshore waters along the continental slope off Nova Scotia, Canada (O'Brien and Whitehead, 2013). In 2006, their status under the Canadian *Species at Risk Act* mandated legal protections, including identification of critical habitat and an assessment of current threats, to aid in ongoing monitoring and recovery measures (DFO, 2016). Their core habitat is concentrated around three submarine canyons along the edge of the eastern Scotian Shelf: The Gully, Shortland, and Haldimand canyons (DFO, 2016). Based on long-term sightings, passive acoustic monitoring, and population analyses, these three canyons, together with the inter-canyon areas, support all the life-history functions (foraging, socializing, reproduction) for NBW on the Scotian Shelf (Stanistreet et al., 2021). Since the end of commercial whaling, threats considered a high risk to NBW recovery include entanglement in fishing gear, ship strikes, military sonar, potential oil spills, and climate change. (DFO, 2022a; Feyrer et al., 2021; Hooker et al., 2019; Whitehead and Hooker, 2012). Beaked whale species (i.e. Ziphiidae sp.) have been documented to ingest plastics at a high rate (Eisfeld-Pierantonio et al., 2022; Kühn and van Franeker, 2020), and plastic has previously been reported as part of NBW stomach content analysis in other areas (Benjaminsen and Christensen, 1979; Fernández et al., 2014). However, the magnitude of the problem for the Scotian Shelf population is currently unknown. Determining the extent of exposure to plastic debris is necessary to assess the potential prevalence of plastic ingestion and whether it may impact the recovery of Scotian Shelf NBWs.

In the 1990s, Dufault and Whitehead (1994) examined the amount of floating plastic debris in The Gully and surrounding area. Despite its offshore location, The Gully contained higher average abundances of both small and large floating plastic debris compared to the surrounding continental slope areas (Dufault and Whitehead, 1994). Large floating debris, such as grocery bags, nylon rope, Styrofoam, and pieces of commercial packaging, were found in greater density in The Gully than the North Pacific, North Sea, and Mediterranean Sea over a similar period (Dufault and Whitehead, 1994). High rates of plastic litter observed on nearby Sable Island beaches in the mid 1980s was also consistent with the levels of debris in this adjacent marine area (Lucas, 1992). Plastic sources identified by the authors included the shipping and fishing industries, as well as recreational vessels and oil rigs, with most items originating from within Canada (Dufault and Whitehead, 1994; Lucas, 1992). In the intervening decades, several changes in marine regulations and human activities in Canada have altered the anthropogenic seascape of these offshore waters, potentially influencing the amount, character, and/or quantity of plastic debris in the region. In the early 1990s, due to the collapse of commercial cod stocks, a number of fisheries were shut down across the Scotian Shelf (Myers et al., 1997). Then in 2004, The Gully was established as a Marine Protected Area (MPA) under *Canada's Oceans Act* (2001), which restricted fishing and vessel activity in a small deep-water area (475 km^2) of "Zone 1". However, long-line fishing activities have continued in the outer zones of the MPA, within the nearby Shortland and Haldimand canyons, and in the adjacent shelf edge waters (Butler et al., 2019). From 1999 to 2018, the Sable Offshore Energy Project operated five natural gas extraction facilities and discharged produced water ~45 km eastward of The Gully (CNSOPB, 2022; Niu et al., 2016). In the last fifteen years, several organizations in regional proximity of The Gully MPA have initiated programs to support responsible disposal of garbage from marine sectors, such as the Ship to Shore program, the Fishing Gear Coalition of Atlantic Canada, and Debris Free Fundy. It is unknown whether and how the levels of plastic debris reflect the changes in human activities in this area over the intervening period.

With the creation of the no take zone within the MPA, The Gully now provides a small offshore oasis of reduced human impact (Feyrer et al., 2021). This conservation area is however, surrounded by areas of human use along the shelf and slope for fishing, oil and gas exploration, and shipping (DFO et al., 2005; Rozalska and Coffen-Smout, 2020). The potential consequences of plastic pollution for Scotian Shelf NBWs from the variety of contemporary sources adjacent to their critical habitat are currently unknown. The limited geographic range and small size of this NBW population may make them more vulnerable to the range of potential impacts from ingested plastic pollution. In addition, mortalities from plastic consumption (e.g., linked to blockages), can have significant demographic impacts on small isolated populations, with the potential to undermine the persistence of this species (Eisfeld-Pierantonio et al., 2022). In this paper, we expand on the work begun by Dufault and Whitehead (1994) to estimate the current levels of marine debris in the region and characterize the composition of plastics in NBW critical habitat. We replicate the original study methods and include new investigative tools to examine the change over time in both large and small plastic debris, and finally, compare the composition of environmental plastics to those found in the stomach contents of three NBW stranded in the region. Our study provides rare longitudinal data on a persistent environmental pollutant in an understudied region of the global ocean. Further, understanding linkages between microplastic persistence in the environment and ingestion in the Scotian Shelf NBW population is vital information for species managers. Trends in marine plastic debris are also important to consider when assessing the full life cycle of risks and environmental impacts of human activities and evaluating the efficacy of conservation areas at mitigating threats to conservation targets.

2. Methods

2.1. Study area and sample design

Plastic debris was sampled from three canyons along the edge of the Scotian Shelf, 200 km east of Nova Scotia, Canada: The Gully, Shortland Canyon, and Haldimand Canyon (Fig. 1). The Gully is the largest submarine canyon in the western North Atlantic, at 40 km long, 16 km wide, and over 2500 m deep. To the east, Shortland canyon (27 km long) and Haldimand canyon (20 km long) are smaller and only extend 1000 m deep (Table 2 in Moors-Murphy, 2014). The deep-water areas (>500 m) of these three canyons are recognized as critical habitat for the Endangered Scotian Shelf population of northern bottlenose whales (DFO, 2016).

Debris surveys were conducted over multiple years in two different periods: the 1990s ("historical") and the 2010s ("contemporary") (Table 1). Over this timeframe, three different methods were employed that targeted different size classes and types of plastic debris, although not every method was used in every year (Table 1). Data were collected

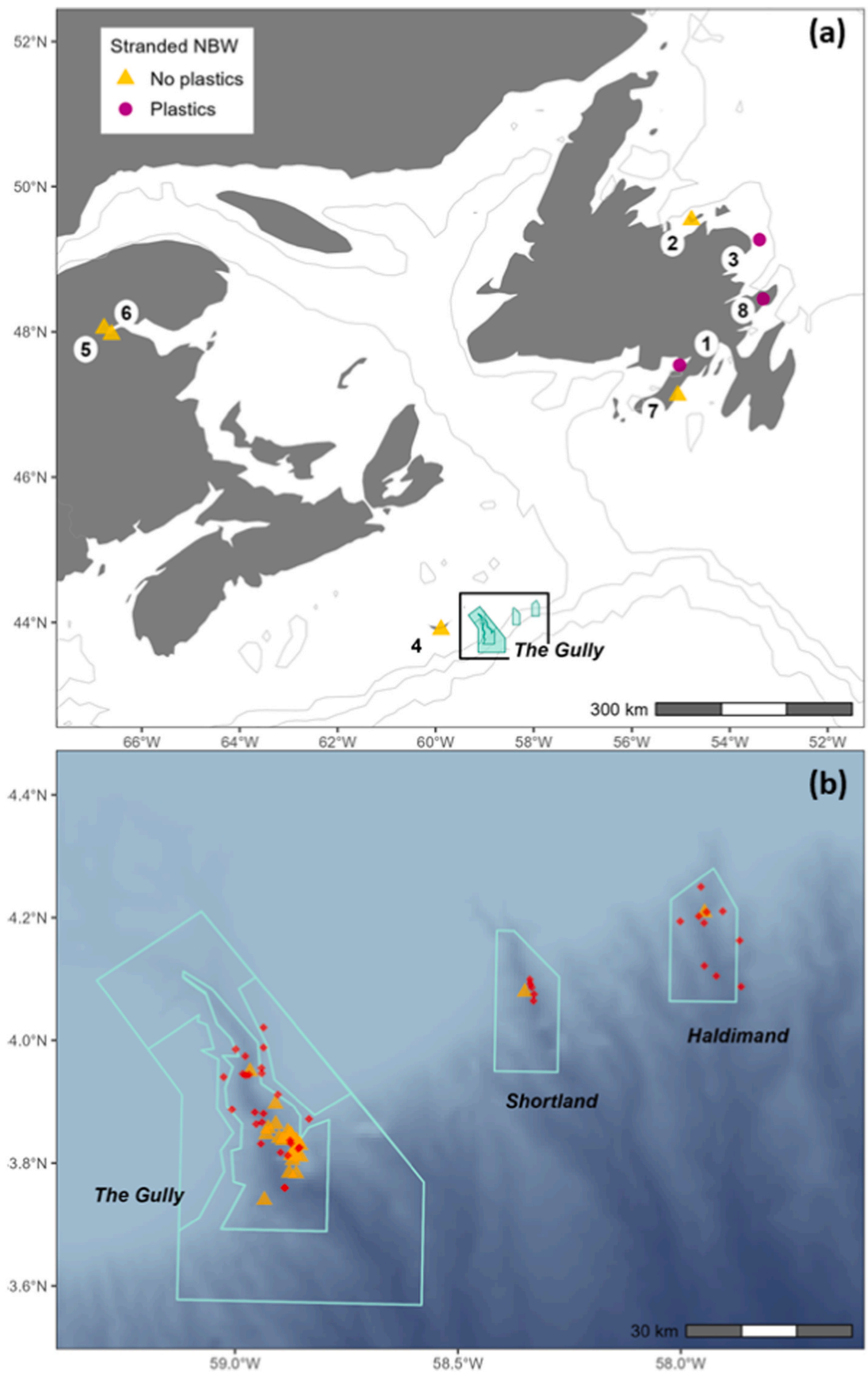

Fig. 1. (a) Location of The Gully in relation to Atlantic Canada, western North Atlantic. The locations of stranded northern bottlenose whales (NBW) are denoted by numbered points for whales with plastics (purple circles) or without plastics (yellow triangles) in their stomachs. (b) Outline of critical NBW habitat (light blue polygons) in The Gully, Shortland, and Haldimand canyons and the locations of visual surveys (orange triangles) or neuston tows (red diamonds) conducted in 2015 and 2016.

during cetacean focused research trips to the canyons typically during July and August. In 1990, surveys were conducted on board a 10 m sailing vessel (*Elendil*); all remaining years sampling was conducting aboard a 13 m Valiant 40 sailing vessel (*Balaena*). The sampling and laboratory processing methods, which were consistent across time and originally detailed by Dufault and Whitehead (1994), are briefly described below. Differences in laboratory processing from the original methods undertaken for contemporary samples are noted.

2.1.1. Visual surveys for large debris

Visual surveys were conducted by one observer who stood forward of the mast and scanned the surface of the water from the bow to 90°, either port or starboard (after Day and Shaw, 1987). Items observed within 15 m were recorded, and distance from the boat was estimated. Surveys typically followed a single bearing of travel, lasting 30–40 min but were occasionally shortened (to ~20 min) due to whale sightings. Spatial coordinates were recorded at the start and finish of each survey, and when possible every 15 min. Positional data were used to calculate survey transect length and surface area (i.e. 15 m strip width * survey length) for abundance calculations. Whenever possible, surveys were paused, and large debris was brought on board for detailed identification. Occasionally, poor visibility due to fog, mist, and cloud cover was

Table 1
Summary of sampling conducted for plastic debris analyzed in this study. Sample types, year, time period, number of samples in each category, and related references. Values in cells are number of replicates taken for each year and sampling method. Size class sampled by each method in brackets. NBW = northern bottlenose whale. See methods for further details.

Period	Year	Sample types				Reference
		Bulk (μm – mm)	Neuston tow (μm – cm)	Visual survey (cm – m)	NBW stomach contents (μm – m)	
Historical	1990	–	25	20	–	Dufault and Whitehead (1994) Whitehead, unpublished
	1996	–	–	8	–	Whitehead, unpublished
	1997	–	10	16	–	Whitehead, unpublished
	1999	–	10	10	–	Whitehead, unpublished
Contemporary	2015	–	26[a]		–	This study
	2016	–	–	22	–	This study
	2019	15[a]	16[a]	–	1[a]	This study
	2021	–	–	–	1[a]	This study

[a] Subset of particles sent for FTIR spectroscopy.

encountered, which would bias abundance estimates downwards. The visual survey protocol was identical across all years.

2.1.2. Neuston net tows for small and micro debris

A neuston net (0.4 × 0.4 m opening, 308 μm mesh size) (Sameoto and Jaroszynski, 1969) was towed alongside the boat outside of the wake at ~3 knots for approximately 1 nautical mile (1.85 km). Boat speed, sea state, and spatial coordinates (latitude and longitude) were recorded at the start and end of each tow. After a 20 min towing period, the net was hauled on board and the cod end removed. Contents were rinsed with seawater into a clean bucket, and obvious large organic material (e.g., algae, crustaceans, cnidarians, etc.) was carefully removed. The remaining contents were strained through a paper filter. The bucket was thoroughly rinsed to ensure all items were transferred to the filter. The filter was then folded closed, placed in a resealable plastic bag, and frozen for transport to the laboratory.

In the laboratory, filters were rinsed with freshwater and dried in an oven (at 35–50 °C). Coffee filters were then cut open and visually examined under a dissecting microscope at 40–100× magnification. Suspected plastic particles were removed by hand and placed in separate jars (historical samples) or on double-sided tape in Petri slides (contemporary samples). For contemporary samples, all suspected plastic particles were counted and photographed using a microscope-mounted digital camera (Nikon SMZ-25 fitted with a Di-3 digital camera, Nikon Instruments, Inc.). All particles were classified according to colour and morphology following standard criteria (Hartmann et al., 2019; Hidalgo-Ruz et al., 2012; Rochman et al., 2019; see Supplementary Materials Table S1-1 for definitions used). Particle size (for 2019 samples only), measured as the longest dimension of each particle, was determined from the microscope photographs using image analysis software (ImageJ, version 1.52 h; Rasband, 2018). For historical samples, data were reported as total number of items per surface area covered by the neuston tow; numbers of particles in different morphology classes or sizes were not recorded and are not available for comparison.

2.1.3. Bulk samples for micro debris

As neuston net tows can underestimate the abundance of smaller-sized microplastics (<300 μm) present in seawater samples (Covernton et al., 2019), an additional sampling method was used to target micro-sized classes of particles and to supplement the information collected via neuston tows for the 2019 sampling year. Bulk surface water samples were collected at either the start or end of neuston tows using a canvas bucket and natural fiber rope to fill pre-rinsed 1 L glass jars (after Barrows et al., 2017; Green et al., 2018). Bulk samples were transported back to the laboratory and stored at room temperature until processed. In the laboratory, samples were vacuum filtered under a laminar flow hood onto 8 μm, 47 mm diameter polycarbonate membrane filters (PCTE, Sterlitech). Lids, jars, and sides of vacuum funnels were rinsed three times with ultrapure water (Milli-Q) to ensure all particles were transferred onto the filter. Filters were carefully removed and transferred to Petri slides then dried (covered) in a desiccator. All suspected particles were identified, counted, sized, and categorized as for neuston tow methods.

2.2. Stomach contents from northern bottlenose whales

Recent strandings of NBWs in Atlantic Canada (Fig. 1a) presented the opportunity to examine whales' stomach contents for plastic debris and qualitatively compare materials with samples collected in their critical habitat. When possible (based on the condition of the carcass and state of internal organs), the stomach ($n = 5$ whales) was collected, opened, washed in a bucket of clean saltwater collected on site, again with fresh water at the stranding network head quarters, and contents were sieved and dried. Of the three stranded NBW that were found in an advanced stage of decay, only one was necropsied, with the stomach being visually assessed for plastics, and contents removed by hand. Overall, three out of six NBWs were found to have plastics in their stomachs during routine necropsies (Table 2). Stomach contents containing plastics (Whales 1 and 3, Table 2) were bagged and submitted for laboratory analysis. A single glove from Whale 8 was collected and photographed but was not retained (see Supplementary materials S1, Fig. S1-3). The condition and measurements of these animals are further described in McAlpine et al. (2023) and stranding reports (Ledwell et al., 2020, 2021). While it is possible that some microplastic contamination was introduced along the chain of the collection protocol, the overall sieving methodology was focused on retaining larger plastics which were easily distinguished from airborne contamination. Consequently, it is most likely that microplastics (≤1.5 mm size fraction) reported from stomach contents are underestimated.

In the laboratory, dried stomach contents were removed from the bags and the largest pieces placed in metal trays. These pieces were photographed using a digital camera (Olympus TG-5) mounted on a tripod, and size (as longest dimension) measured with a metal ruler to the nearest 0.5 mm. Since the larger pieces of debris (i.e. ropes and nets) were matted or tangled, they were carefully eased apart by hand with the aid of dissecting scissors and forceps, to separate the individual strands and extract smaller debris items that were either inside or wrapped around the larger pieces of debris (i.e. fragments, films, threads). When needed, the remaining contents from the bags were emptied into a stack of stainless steel sieves (4750, 2000, 1000, 500, 250 μm) to facilitate sorting of the debris. The sieve stack was manually shaken twice for 2 min each. Items retained on each sieve and the receiving pan were then transferred to individual metal trays or glass Petri dishes. Items >4750 μm were inspected by eye and photographed; colour and morphology were recorded, and size (as longest dimension) was measured with a ruler. All sorting and photography was conducted in a laminar flow hood cabinet. The smaller items (≤2000 μm) were

Table 2
Necropsy assessment of stomach contents of NBW stranded across the region. Stomach content assessment depended on the condition and size of the animal. See Supplementary Materials S1 Figs. S1-2, S1-3 for images of plastics collected from Whales 1, 3, and 8. ND = Not determined; Brackets after location names indicate Provinces (NL = Newfoundland and Labrador; NS = Nova Scotia; QC = Quebec).

Whale	Date	Location	Coordinates	Sex	Necropsy	Stomach contents	FTIR
1	August 11, 2019	Harbour Mille (NL)	47.59°N, 54.88°W	F	Y	Plastics, rope, squid beaks, fish otoliths, lenses	Y
2	February 16, 2021	Boyds Cove (NL)	49.47°N, 54.75°W	F	Y	Squid beaks	–
3	March 7, 2021	Musgrave Harbour (NL)	49.25°N, 53.51°W	F	Y	Plastic rope, plastic fibers, squid beaks	Y
4	March 15, 2021	Sable Island (NS)	43.93°N, 59.97°W	F	N	–	–
5	September 30, 2021	Point-à-la-Croix (QC)	48.01°N, 66.73°W	M	Y	Squid beaks, fish eye lenses	–
6	September 30, 2021	Point-à-la-Croix (QC)	48.01°N, 66.73°W	F	N	–	–
7	October 1, 2021	Mortier Bay (NL)	47.16°N, 55.14°W	M	Y	Squid beaks	–
8	June 12, 2022	Knights Cove (NL)	48.54°N, 53.32°W	ND	Y	Nylon glove, squid beaks	–

visually observed under a dissecting microscope fitted with a digital camera at either 10X (for the 2000 and 1000 μm fraction) or 50× (remaining sieve size fractions) magnification; suspected plastic particles were counted, and colour and morphology recorded (using the same categories as for surface water samples), then photographed. Size (as longest dimension) for the smaller debris was measured using image analysis software (Image J) as for previous samples.

2.3. Polymer identification via FTIR

Polymer composition for select particles from contemporary samples (bulk, neuston tow) and NBW stomach contents were identified by Fourier Transform Infrared (FTIR) spectroscopy at Surface Science Western, University of Western Ontario, London, Ontario, Canada. Following Dimitrijevic et al. (2019), we randomly selected a subset of particles, aiming for 10% of particles from each major colour and morphology combination (i.e. red fibers, green fragments, black films, etc.) per sampling method. If a large debris item was selected (mostly from NBW stomachs), a smaller section (approximately 2 cm × 2 cm) was removed and sent for FTIR.

Suspected plastic particles were transferred to a diamond compression cell and analyzed by FTIR in transmission mode under the Hyperion 2000 microscope attached to a Bruker Tensor II spectrometer. Items larger than 2 mm × 2 mm were analyzed using a Platinum attenuated total reflectance (Pt-ATR) attachment. For microfibers that had FTIR spectra consistent with a cellulosic material, the method of Cai et al. (2019) was applied to distinguish between natural and semi-synthetic cellulosic materials. Each spectrum in question was visually examined for a peak, shoulder, or no peak at ~1105 cm^{-1}. Particles with spectra that contained a peak at this wavelength were classed as natural cellulosic fibers (see Data Analysis section below); particles with no peak or a shoulder at this wavelength were classed as semi-synthetic fibers. Thus, for the purposes of this study, we define natural cellulosic fibers as non-plastic particles made from natural sources but have been manipulated for human purposes (e.g., textile fibers made from processed and dyed cotton or linen), while semi-synthetic cellulosic fibers are particles comprised of rayon or viscose (and considered plastics).

2.4. Contamination reduction protocols

The contamination of microplastics from airborne or water sources has only recently been considered. As such, few contamination prevention protocols were implemented for the historical sample processing. One of the largest risks is the potential for airborne contamination during laboratory processing (N.K., unpublished observation, Woodall et al., 2015). However, as airborne contamination from indoor air is typically fibers in dust (e.g., Catarino et al., 2018), and most of the plastics recovered in the 1990s were readily distinguishable from these types of particles, we believe any potential contamination for the 1990s samples was likely to be small to negligible.

For contemporary samples, consistent efforts were made to minimize sample contamination in the field and laboratory (following Dimitrijevic et al., 2019). In the field, researchers wore clothing made from natural materials (i.e. not fleece) and minimized the amount of time samples were exposed to air during the collection process. In the laboratory, all materials and instruments used were made of metal (stainless steel or aluminum) or glass (except for squirt bottles made of polyethylene), and all were rinsed 3 times with ultra-pure water (0.22 μm membrane filtered deionized water; Milli-Q Synergy, Millipore) prior to use. A 100% cotton lab coat or Tychem 2000 (Dupont) coveralls, as well as a muslin cloth head scarf and nitrile gloves, were worn for all laboratory work, and all clothing was lint rolled prior to the start of any processing. Microscope covers were taped in place (after Woodall et al., 2015) to protect the samples from airborne contamination. A laminar flow hood cabinet was also used for sample sorting or processing when not under the microscope. All work areas and equipment were cleaned three times with ultra-pure water prior to opening and/or processing samples.

Background and procedural blanks were also collected during laboratory processing of all contemporary and NBW stomach samples. For the bulk samples, one background blank was created for each sample, and consisted of a polycarbonate membrane filter placed in a Petri slide, which were opened whenever the samples were exposed to air. Three procedural blanks were conducted, and processed using identical laboratory methods as samples, except 1 L glass jars were filled with ultrapure water instead of seawater. For the neuston tow samples, six background blanks were used for each sampling year (i.e. $n = 6$ in 2015 and $n = 6$ in 2019), prepared in the same manner as for the bulk samples. It was not possible to conduct procedural blanks for neuston tows. For the NBW stomach samples, a polycarbonate filter was placed within a glass Petri dish on each sample processing date, for a total of 10 background blanks ($n = 7$ for Whale 1 and $n = 3$ for Whale 3). All blanks were analyzed under a dissecting microscope, and particles classified by type, colour, and size, in the same manner as for actual samples.

2.5. Data analysis

All calculations and statistical analyses were conducted using the R software environment (R Core Team, 2022) in RStudio (version 2022.02.2).

For contemporary samples, particles found in blanks were used to correct the final particle counts by establishing a limit of detection (LOD), following Waddell et al. (2020) and De Witte et al. (2014). The LOD was calculated as the mean + 3 * standard deviation (SD) for each particle colour and morphology combination. Background and procedural blanks were combined and treated equally for the calculation, which was applied within each sampling method (i.e. we created LODs separately for the sets of particles extracted from bulk and neuston tow sampling methods). Corrective action (subtraction of the LOD from the number tallied in the sample) was only taken if an item of the same colour and type was found in both the blanks and the samples. If the sample totals were < LOD (i.e. subtraction resulted in a negative number), the value was set to zero.

Following blank correction of contemporary samples, particle counts per sample were further adjusted following the results of FTIR spectroscopy to report a final plastic particle count (Coverton et al., 2019; Huntington et al., 2020). The number of particles in each colour-morphology combination were multiplied by a correction factor to account for the amount of visual identification error as identified by FTIR spectroscopy (i.e. if 10 blue fibers were analyzed by FTIR and five were confirmed as plastic, particle counts were multiplied by 5/10) (Coverton et al., 2019). This plastic particle count includes plastics as well as semi-synthetic fibers, and paint containing alkyds. Final counts were rounded up to the nearest whole integer (to maintain nature of count data) then standardized by sample volume (bulk) or surface area (neuston tows). For blank correction of NBW stomach samples, the smaller sample size allowed us to adjust final particle counts by directly subtracting particles in procedural and background blanks of the same colour and morphology from their corresponding sample (Battaglia et al., 2020).

We examined the change in large and small plastic debris over time in two ways. While samples were collected from The Gully in every year, samples from Shortland and Haldimand canyons were not. Thus, comparisons across time were analyzed only for samples collected from The Gully in order to match the historical data available (Table 1). First, we examined change across years using Kruskal-Wallis non-parametric ANOVA, followed by Dunn's post-hoc test, as these data did not meet parametric test assumptions of normality nor homogenous variances (even after transformation, as determined by Shapiro-Wilk's and Levene's tests). We adjusted p values (p_{adj}) to account for multiple comparisons with the Benjamini-Hochberg method to control the false discovery rate. These analyses were carried out using the *stats* package, and the *dunn_test* function in the *rstatix* package (Kassambara, 2021). Second, we combined data into periods – contemporary (2010s) and historical (1990s) – and tested the difference in plastic abundance between periods using a non-parametric Wilcoxon rank sum test. These tests were conducted separately for neuston tows and visual surveys. For all tests, significance was assessed at $\alpha = 0.05$. Comparisons of contemporary plastic particle concentrations among canyons for each sampling method (bulk, neuston tow, visual survey) are presented in Supplementary Materials S2.

To examine the potential for NBWs to ingest plastic debris from their critical habitat, we made qualitative comparisons of the plastic debris sampled in seawater to debris found in NBW stomach contents. We visually compared the size ranges and polymer types of particles sampled in 2019 (by bulk and neuston tow) to those removed from NBW stomachs in 2019 and 2021. All particles identified by FTIR that were sampled using bulk or neuston tows were pooled across canyons within sampling method where applicable to represent plastics removed from seawater in NBW critical habitat. For these comparisons, we also included natural cellulosic material (which were mostly fibers); although not plastics, these particles often contain colorants (dyes, pigments) and chemical additives (flame retardants, antimicrobial agents, formaldehyde), and may be equally harmful to marine biota as plastics (reviewed in Athey and Erdle, 2022).

3. Results

3.1. Historical vs. contemporary periods

In The Gully, small plastic debris sampled using neuston net tows significantly increased over time (Kruskal-Wallis $H(X^2) = 36.7$, $df = 4$, $p < 0.0001$) (Fig. 2a). Significantly higher median values of small plastics (no. km^{-2}) were observed in 2015 and 2019 than in 1990 ($p_{adj} < 0.001$) and 1999 ($p_{adj} < 0.05$). There was no significant difference in values between 2015 and 2019 ($p_{adj} = 0.8$). Values in 1997 were intermediate between these two groups, being significantly greater than in 1990 ($p_{adj} < 0.01$), but not significantly different from values in 1999 after correction for multiple testing ($p_{adj} = 0.076$).

Large plastic debris sampled using visual surveys significantly decreased over time (Kruskal-Wallis $H(X^2) = 25.3$, $df = 4$, $p < 0.0001$) (Fig. 2b; Supplementary Materials S2 Table S2-1). While median abundances in the 1990s were variable among years, there were significantly lower median values of large plastic debris observed after 1996 (all $p_{adj} < 0.01$).

Comparing across periods, the abundance of small plastic debris was significantly greater in the 2010s than the 1990s (Wilcoxon W = 120, $p < 0.0001$) (Fig. 3a). In contrast, there was a significant decrease in large plastic debris from the 1990s to the 2010s collected via visual surveys (Wilcoxon W = 774.5, $p = 0.0012$) (Fig. 3b).

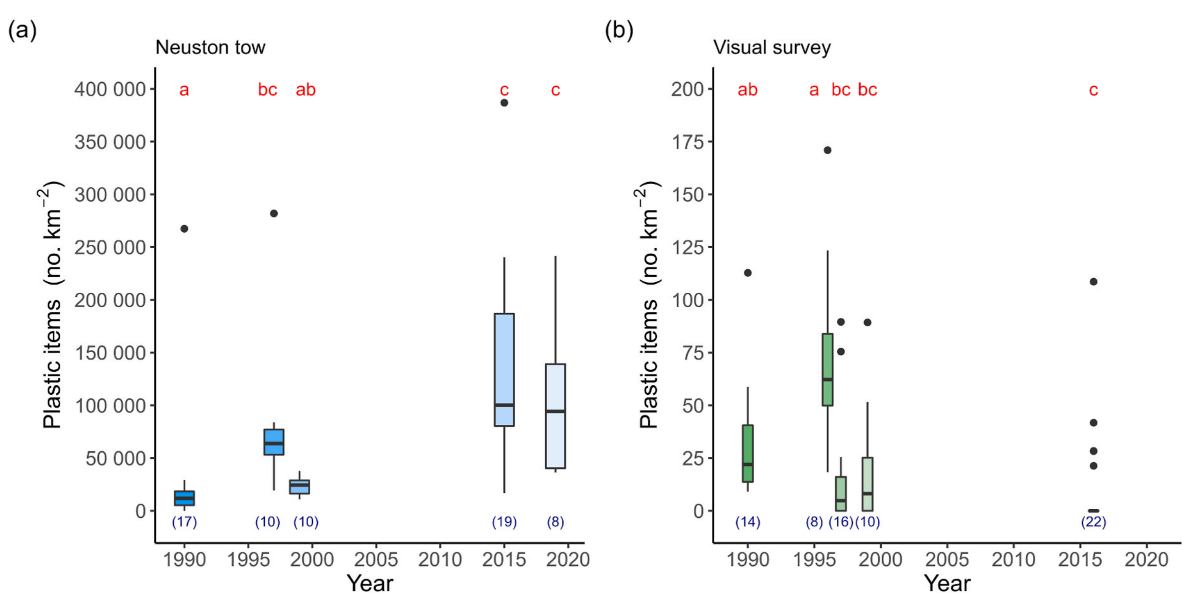

Fig. 2. Plastic abundances in The Gully from 1990 to 2019 for (a) small plastic debris sampled via neuston net tows, and (b) large plastic debris observed in visual surveys. Numbers in parentheses are sample sizes within each year. Different red letters denote significant differences (p < 0.05) among years. Boxes show the interquartile range (IQR: 25th to 75th percentile), the solid black line represents the median, the whiskers represent the minimum and maximum values or 1.5*IQR from the first or third quartile, whichever is less. Circles represent outlying values which are larger or smaller than 1.5*IQR.

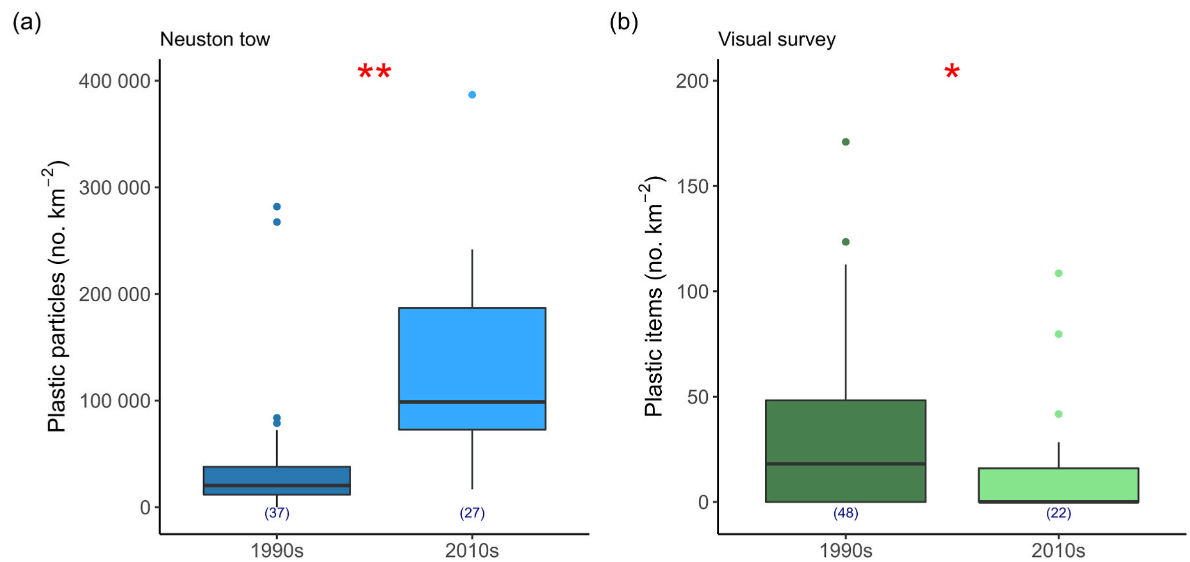

Fig. 3. Plastic abundance in The Gully sampled in historical (1990s) and contemporary (2010s) periods for (a) small debris sampled using neuston net tows; and (b) large debris observed in visual surveys. Values in brackets are sample sizes within each period. Boxplots with interquartile ranges as in Fig. 1. Asterisks denote significant differences between periods at $p < 0.001$ (**) or $p < 0.01$ (*).

3.2. Contemporary analysis

Plastics and other anthropogenic particles were found in all bulk samples (2019), in all neuston tows (2015 and 2019), and in 5 of 21 visual surveys in 2016. After accounting for contamination and identification errors in bulk and neuston tow samples, total plastic particle numbers were recorded as 246 (bulk 2019), 2970 (neuston tow 2015), and 886 (neuston tow 2019). A total of 14 large items were observed across all visual surveys in 2016.

The concentration or abundance of floating plastic debris was spatially heterogeneous (i.e. patchily distributed across space), varying by 1–3 orders of magnitude, depending on sampling method, year, and canyon location (Table 3, Supplementary Materials S2 Fig. S2-1a). Micro-debris concentrations, as sampled using bulk methods, ranged from 3 to 30 microplastics L^{-1}, and were comprised primarily of fibers (Supplementary Materials S2 Figs. S2-2). Small and micro-sized plastic debris abundances, as sampled using neuston tows, ranged from 5586 to 438 196 plastics km^{-2} and captured a greater diversity of plastic morphologies (fibers, fragments, films, flakes, spheres) (Supplementary Materials S2 Figs. S2-1b, S2-2). Large debris abundance ranged from 0 to 108.6 items km^{-2} (Supplementary Materials S2 Figs. S2-1c). Large debris sampled via visual surveys consisted of plastic sheets, plastic grocery bags, food wrappers, a yogurt container, small plastic fragments, rope, and fishing buoys (Supplementary Materials S2 Table S2-1).

When compared to studies employing similar methodologies, micro, small, and large debris estimates from The Gully and area canyons are greater (in all but one case) than those reported in other surface waters across the North Atlantic (Table 3).

Of the 596 particles sampled from bulk and neuston tows analyzed with FTIR spectroscopy, 81.4% were plastic, another 6.5% were identified as other anthropogenic material (i.e. glass, metal, rubber), 10.6% were organic (plant material, hair, wool, etc.), and 1.5% were unable to be identified (no match in FTIR library). A greater diversity of polymer types were sampled with neuston tows than bulk methods (Fig. 4). The most common polymer types found using bulk methods were fibers of polyethylene terephthalate polyester (PET) and natural cellulose. The most common polymer types collected by neuston tows were polyethylene (PE), polypropylene (PP), PET, and natural cellulose. Most natural cellulose, PET, nylon, and acrylic particles were fibers (Fig. 4a), while PE and PP were more frequently found as fragments (Fig. 4b). Less frequently encountered polymer types included paint flakes containing alkyds, semi-synthetic cellulose, and other plastics (e.g., polyvinyl chloride, polystyrene, and ethylene vinyl acetate). Sizes of particles sampled using the bulk method ranged from 35 μm to 12.5 mm (median size = 744 μm), while debris sampled by neuston tows ranged from 90 μm to 39.1 mm (median size = 1207 μm; Fig. 5). The size distributions of fibers (Fig. 5a) were similar for both bulk and neuston tow samples, although fragments sampled in neuston tows tended to be larger than those sampled using bulk methods (Fig. 5b).

3.3. Debris characterization in NBWs and their critical habitat

Prey consumed by Whale 1 were preliminarily identified as *Histioteuthis* from squid beaks, and redfish (*Sebastes* sp.), Atlantic cod (*Gadus morhua*), and long finned hake (*Phycis chesteri*) from otoliths; additional bones and eye lenses were not speciated. Differing amounts of plastic items were collected from the stomachs: 384 distinct plastic items were found in Whale 1, while 32 items were found in Whale 3. Most plastic items were biofouled or colonized by fauna (i.e. worms, small crustaceans), suggesting residency in the water column or at the surface for some time prior to ingestion. Smaller plastic pieces were often found entangled within the larger items (i.e. within mats or tangled fragments of rope), but it is unknown if this occurred within the stomach or in the ocean prior to ingestion.

Multiple item morphologies were observed in the NBW stomachs: Large items (>2.5 cm) included fragments of fishing nets, ropes, bottle caps, cups, and food wrappers (similar to items observed during visual surveys in 2015 and 2016; Supplementary Materials S2 Table S2-1), while smaller items consisted of rope filaments (threads), smaller plastic fragments, fibers, and flakes (similar to particle morphologies sampled in neuston tows) (Supplementary Materials S1 Figs. S1-2, S1-3). The majority of particles in Whale 1 stomach (96%) were fibers or fibrous, from nets or ropes (as whole pieces, sections of rope or net, large filaments, or individual fibers); the remaining 4% of particles were fragments. We found only fibers (ropes, nets, filaments, fibers) in Whale 3's stomach, with 84% of all plastic material recovered (by weight) was from a single large piece of matted rope (Supplementary Materials Fig. S1-2a). Items removed from NBW stomachs spanned the widest range of sizes, from 150 μm to 1.13 m (median size = 15 mm) for Whale 1, and 800 μm to 20 cm (median size = 21.2 mm) for Whale 3 (Fig. 5).

Of the 85 particles from the NBW stomachs analyzed by FTIR, 94.7%

Table 3
Concentration or abundance of items of micro-sized, small and large plastic debris found in this study and comparisons to studies conducted in the North Atlantic using similar sampling methodologies. NWA = Northwest Atlantic.

Location	Year	Method	Mean ± SD (n)	Source
Micro debris (no. L^{-1})				
The Gully	2019	Bulk (1 L)	19.6 ± 7.5 (7)	This study
Haldimand canyon	2019	Bulk (1 L)	17.7 ± 6.35 (3)	This study
Shortland canyon	2019	Bulk (1 L)	11.2 ± 6.3 (5)	This study
Maine coast, USA	2014	Bulk (1 L)	5.9 ± 4.4 (17)	Barrows et al. (2017)
Plymouth Sound, UK	2015	Bulk (1 L)	2.6 [5.42][c] (10)	Green et al. (2018)
Micro and small debris (no. km^{-2})				
The Gully	2015	Neuston net (308 μm)	136 832 ± 93 183 (19)	This study
Haldimand canyon	2015	Neuston net (308 μm)	194 175 ± 137 643 (7)	This study
The Gully	2019	Neuston net (308 μm)	112 017 ± 81 027 (8)	This study
Haldimand canyon	2019	Neuston net (308 μm)	40 354 ± 8710 (3)	This study
Shortland canyon	2019	Neuston net (308 μm)	21 534 ± 21 259 (5)	This study
The Gully (station GULD03)	2016	Manta net (200 μm)	23 370	de Mendonça et al. (2021)
The Gully	2016	Neuston net (300 μm)	423.59 ± 522.10 (4) [a]	Rivers et al. (2019)
NWA (44 °N)	1986–2008	Neuston net (335 μm)	3659 ± 1400	Law et al. (2010)
Gulf of Maine	1986–2008	Neuston net (335 μm)	1534 ± 200	Law et al. (2010)
Coastal Nova Scotia (Lunenburg)	2018	LADI net (335 μm)	4905.06 ± 4866.21 (6)	Smith et al. (2022)
Scotian Shelf (stations HL02, HL03, HL04)	2016	Manta net (200 μm)	9580–28 186	de Mendonça et al. (2021)
Continental Slope (stations RS01, HL06.3)	2016	Manta net (200 μm)	14 669–58 788	de Mendonça et al. (2021)
NWA (Gulf Stream Frontal Area 2 station)	2016	Neuston net (300 μm)	4338.15 ± 7293.28 (4)[a]	Rivers et al. (2019)
Maine coast, USA	2014	Manta net (335 μm)	213 709[b] (6)	Barrows et at. (2017)
Large debris (items km^{-2})				
The Gully	2016	Visual survey	10.4 ± 25.1 (22)	This study
Portuguese coastal zone	2011	Visual survey	2.98 (1.98–4.48 95% CI) (13)	Sá et al. (2016)
Azores archipelago	2015–2017	Visual transects	1.39 ± 0.14 (2406)[d]	Chambault et al. (2018)

[a] Reported fragments only.
[b] SD not reported.
[c] Reported as median [interquartile range].
[d] After observer correction.

were identified as plastic, 4% were natural celluloses, and 1.3% were of other anthropogenic origin (identified as an aluminosilicate fragment). Whale 1 had a greater diversity of polymer types than Whale 3, likely reflecting the larger number of items ingested (Fig. 4). The most commonly observed polymers in both whales included PP, PE, PET, and nylon, although the proportions differed between the whales (Fig. 4). Whale 1 had also ingested acrylic and semi-synthetic and natural cellulosic fibers, as well as paint fragments, but in smaller proportions to the other material types.

The polymer types recovered from the NBW stomachs were similar to that sampled from surface waters in their critical habitat in 2019, although the pattern varied between the whales (Fig. 4). Both stomach contents and neuston and bulk samples all had high relative proportions of PP and PE fragments, and PP, PET, and nylon fibers. In contrast, NBW stomach contents contained PE fibers and PET fragments, which were not found in surface water samples (Fig. 4). For both fibers (Fig. 5a) and fragments (Fig. 5b), debris from the NBW stomachs ranged across all size classes (micro, small, and large) and overlapped with the size distributions from the neuston tows and bulk methods that targeted the micro and small size classes. Overall, the polymer type and size distributions of Whale 1's stomach contents (stranded in 2019) most resembled the surface samples collected via neuston tow in 2019 (Figs. 4 and 5).

4. Discussion

Here we present the first long-term study of concurrent change in multiple size classes of plastic debris in the western North Atlantic. Over the last three decades, the smallest size classes of debris have increased, while the largest size classes have decreased. The current abundance of plastic debris in The Gully, a marine protected area and critical habitat of Endangered Scotian Shelf NBWs, is higher than in surrounding oceanic surface waters, and displays a wide variation in size, polymer types, and particle morphologies. This mixture of plastic pollution appears to be derived from fishing gear, commercial uses (food packaging and textiles), recreational activities, and vessels. NBWs are consuming plastic across all size classes of debris (micro to large), consistent with the polymer types and particle sizes collected from surface waters in their critical habitat. While the long-term consequences of plastic ingestion on NBWs is unknown, the current high abundance of plastic debris in their critical habitat, in combination with the increasing abundance of micro-sized and small plastics over time, suggests associated health (e.g., potential 'plasticosis'; Charlton-Howard et al., 2023) and welfare impacts of ingested plastics should be accounted for in future recovery plans (Eisfeld-Pierantonio et al., 2022).

The Gully and area canyons appear to be a regional hot spot of floating plastic debris in the western North Atlantic. Although abundances are not as high as levels found in some urbanized coastal environments (e.g., Mediterranean (van der Hal et al., 2017), Hong Kong (Cheung et al., 2018)), the abundance of small plastic debris measured in contemporary samples in The Gully and area canyons is higher by 1–2 orders of magnitude than what has been measured in the surrounding continental shelf areas (Law et al., 2010; Rivers et al., 2019; de Mendonça et al., 2021, Table 3). In an earlier study, Law et al. (2010) examined the abundance of floating plastic debris across the western Atlantic, finding that the highest surface concentrations of plastic debris corresponded to convergence areas created by wind-driven Eckman currents and geostrophic circulation. From this analysis, Law et al. (2010) identified a small convergence area in the western North Atlantic which overlaps the geographic area of The Gully and area canyons. Surface circulation in The Gully region appears to be influenced by the presence of anticyclonic (clockwise) gyres that circle the surrounding banks (Sable and Banquereau), while a cyclonic (anticlockwise) partial gyre forms over the channels between the banks (Han, 2003; Kenchington et al., 2014). Particularly for The Gully, exchange between the gyres appears to produce a south-westward flow over the canyon mouth, and a slow north-eastward drift across its head (Kenchington et al., 2020). These gyres may be acting to concentrate or trap floating plastic litter in the area, particularly in summer months when wind forcing and current speeds are typically weakest (Shan et al., 2014a), increasing exposure of NBWs to marine plastic pollution. Sá et al. (2021, 2016) found a similar occurrence in eastern North Atlantic waters off the Portuguese coast, where areas under the influence of dynamic oceanographic features were acting as retention zones, concentrating nutrient,

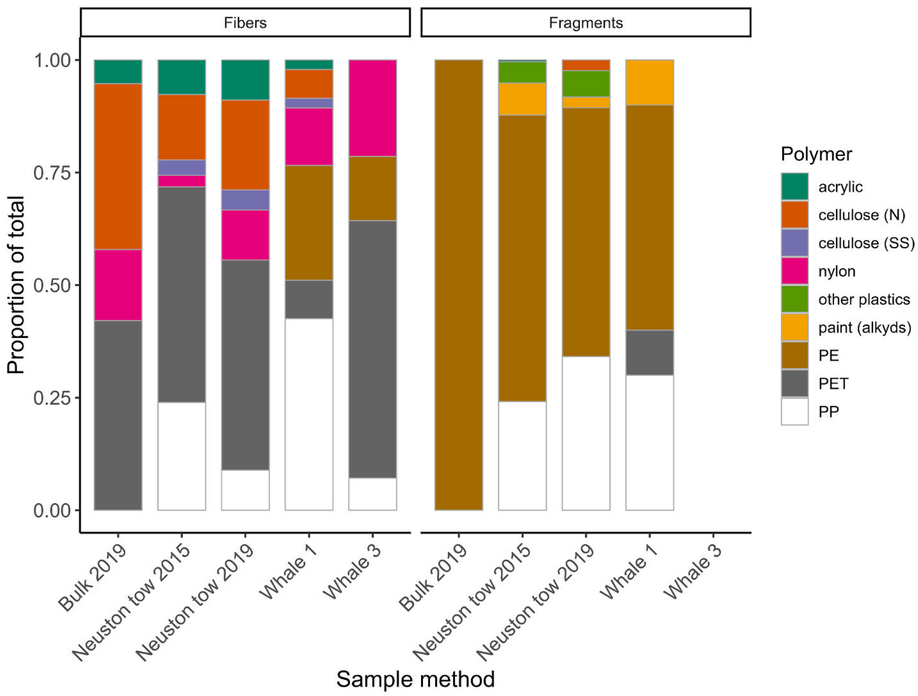

Fig. 4. Polymer composition of plastic fibers or fragments, as determined by FTIR spectroscopy, across bulk and neuston tow sampling methods and from northern bottlenose whale stomach contents. Particles are pooled among canyons for bulk and neuston tow samples. 'Fibers' category includes fibers, filaments, and fiber bundles; 'Fragments' category includes fragments, spheres, films, and flakes. PE = polyethylene; PET = polyethylene terephthalate polyester; PP = polypropylene; 'other plastics' include polystyrene, polyvinyl chloride, and ethylene vinyl acetate. Natural cellulose (cellulose N) represents cellulose particles made from natural sources but have been manipulated for human purposes (e.g., textile fibers made from processed and dyed cotton or linen). Semi-synthetic celluloses (cellulose SS) are particles comprised of rayon or viscose, and were distinguished from natural celluloses following the method of Cai et al. (2019). See methods text for further details. No fragments were recovered from the Whale 3 stomach contents.

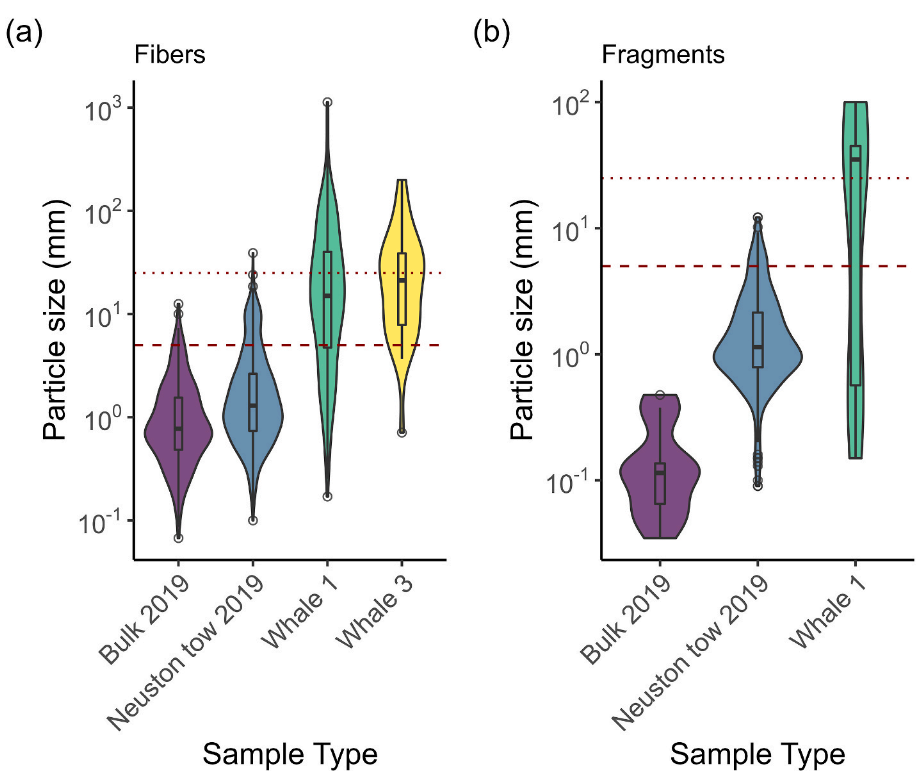

Fig. 5. Size distributions of plastic particles by sampling method or northern bottlenose whale stomach contents for (a) fibers and (b) fragments. Size is length of longest dimension of each plastic particle. Boxplots (with interquartile ranges as in Fig. 1) with outliers (as open circles) are superimposed within each violin. No fragments were recovered from Whale 3 stomach contents. Red dashed and dotted horizontal lines denote 5 and 25 mm size boundaries between micro and small, and small and large size classes, respectively.

plankton, and prey, but also floating marine litter, subsequently increasing interactions of marine mammals with plastic debris.

The diverse nature of plastic products and the multiple routes they can take to enter the marine environment poses challenges in determining their source. The current abundance of plastic debris in The Gully and area canyons displays a wide variation in size, morphology, and polymer type. At present, such an assortment of plastic suggests multiple origins, sourced from consumer products (particularly food packaging and textiles), vessels, industrial activities, fishing, and recreational activities. From field sampling in 2016, de Mendonça et al. (2021) found a lack of plastics on the Scotian Shelf but a higher abundance along the continental slope areas, and suggested a wider oceanic source for plastics in this area. The transport of plastic waste (i.e. food packaging, textiles) from more southern locations (e.g., the North Atlantic subtropical gyre (Law et al., 2010) or along the eastern seaboard of the US) into temperate waters via the Gulf Stream Current could explain higher concentrations of floating plastic in The Gully area despite its offshore location and remoteness from dense centres of human population. Many particles collected using neuston tows in our study were weathered and/or degraded, suggesting a longer residence time in surface waters; however, little data are available on the rates of degradation of plastics in offshore waters (Andrady, 2017). In contrast, the mean annual surface currents in The Gully region are influenced primarily by Scotian Shelf waters flowing south-westward across the shelf-slope boundary (Greenan et al., 2014; Shan et al., 2014a), suggesting that plastics may be transported from the northeast, where there is higher fishing activity (Rozalska and Coffen-Smout, 2020). Given the contemporary diversity of plastic debris sampled, it is possible that the cumulative input from local sources (i.e. fishing and transportation vessels) and long distance transport (from areas of high human population) is converging in The Gully.

While plastic production has dramatically increased, there have been few long-term studies quantifying trends in plastic debris in the marine environment. To date the overall findings have been equivocal, with some detecting increases in the open ocean (Ostle et al., 2019), in bivalves (Halbach et al., 2022), washed up on beaches (Waluda et al., 2020) or sinking to the deep-sea floor (Parga Martínez et al., 2020), while others have failed to detect a significant trend in surface water, biota, or sediments over several decades (Beer et al., 2018; Courtene-Jones et al., 2019). Finding a significant increase in small- and micro-sized plastic debris in The Gully is consistent with the reality that large quantities of plastics continue to enter the ocean annually (Borrelle et al., 2020; Geyer et al., 2017). However, we also observed a concurrent and significant decrease in large plastic debris, suggesting this increase in small- and micro-sized debris may be the result of larger plastics fragmenting into smaller pieces. Macro- and meso-plastics are a major secondary source of microplastics; exposure to UV radiation at the sea surface results in their photodegradation and fragmentation by wave action (Gewert et al., 2015; ter Halle et al., 2016; Weinstein et al., 2016). Alternately, the increase could be the result of long-range transport of small plastic debris from areas of higher concentration that then accumulate in The Gully due to local oceanographic conditions and the long-term durability of plastic materials (Andrady, 2017; Barnes et al., 2009). The decrease in large-size plastics we observed may also be related to regulations implemented between the 1990s and 2010s that eliminated some fishing activities and may have reduced vessel traffic in The Gully. However, this decline in large plastics may also reflect the fact that our visual survey results from the 2010s were dominated by samples taken within a single year, and it is difficult to discern whether 2016 was a true representation of plastic debris within that decade. In contrast, these results do align with subjective observer experiences that large plastic concentrations in The Gully have decreased substantially over the 30 year period (H.W., unpublished observation). While many questions on the cycling, transport, and fate of marine plastics in the ocean remains to be answered (Galgani et al., 2021), plastic production is projected to increase (Lebreton and Andrady, 2019), suggesting plastics, both large and small, will continue to persist in The Gully and adjacent canyons into the near future.

Globally there appears to be an increase in reports of ingestion of marine plastics by cetacean species (Eisfeld-Pierantonio et al., 2022; Fossi et al., 2018), however it is difficult to account for observer bias and increased attention to the issue. While we were only able to opportunistically analyse the stomach contents of two NBWs, the total counts of plastic items recovered were within the range of particles recovered from other beaked whales (Battaglia et al., 2020; Baulch and Perry, 2014). Plastics found in two NBWs killed by whalers in the 1970s suggests ingestion may have been an issue for NBWs since plastics first entered the environment (Benjaminsen and Christensen, 1979). NBW are deep-divers that forage for squid and demersal fish at depths >500 m, and ingest prey through suction feeding (Hooker et al., 2001; Werth, 2006). Given higher ingestion rates observed in beaked and sperm whales (*Physeter macrocephalus*) (Alexiadou et al., 2019; Lusher et al., 2018), it has been suggested that deep-diving cetaceans are more vulnerable to ingestion of plastic debris compared to other coastal or shallower water cetacean species (Eisfeld-Pierantonio et al., 2022). This may be due to their diving and feeding behaviour, rather than the concentrations of plastics within their habitats (Di Beneditto and Oliveira, 2019). In addition to potential interactions with debris in the surface layer, NBW could be encountering plastics trapped or suspended in the deep slow-moving currents within the canyon, particularly in summer months when particles may be retained over several weeks (Shan et al., 2014b). Although some high-density plastics, which are negatively buoyant in seawater, were found in NBW stomachs (i.e. particularly PET, specific density 1.30–1.40 g cm^{-3}), plastic density can be affected by many factors, including changes in particle shape and size, weathering, presence of biofilms, and/or additives (Kooi et al., 2017, 2016). More evidence is required to confirm whether NBWs are also exposed to demersal plastic pollution within their critical habitat.

While we cannot conclusively determine that the plastic debris consumed by NBW was from The Gully or area canyons (nor that plastic was the cause of death), it appears the nature of The Gully and adjacent canyon environments concentrate surface water plastics relative to surrounding areas (Table 3). There is overlap in the morphologies, size classes, and polymer types of plastic debris found in NBW stomach contents and the debris collected from their critical habitat, which is where the majority of the Scotian Shelf NBW population are present year-round (O'Brien and Whitehead, 2013; Stanistreet et al., 2021). Overall, evidence indicates that this population of NBWs are both exposed to and are intentionally or accidently ingesting plastic items within their critical habitat. While more research is required to understand the potential chronic effects on marine mammal health (Zantis et al., 2021), given the large amount of plastic found in the NBW stomachs examined in this study, we agree with Eisfeld-Pierantonio et al. (2022) that ingestion of large, small, and micro-sized plastics are having health and welfare impacts on this Endangered population of NBWs. In future, NBWs may face increased risks from the ingestion of smaller plastic items if these size classes continue to increase in abundance throughout their habitat. Alongside entanglement, vessel strikes, and military sonar, we suggest plastic pollution is an anthropogenic threat posing a high risk to individuals and the recovery of the Endangered SS NBW population (DFO, 2022a; Feyrer et al., 2021).

5. Conclusion

Despite multiple changes in the surrounding anthropogenic seascape over the previous three decades, The Gully continues to sustain high rates of plastic floating debris. Although now recognized as an MPA, the cross-boundary nature of marine plastic debris makes management a challenging problem that transcends action within the boundaries of any protected area. Previous studies on plastic pollution in MPAs highlight the need for preventative measures at the source (Soto-Navarro et al., 2021), as MPA status will not prevent the entry of plastic pollution into a given area (Barnes et al., 2018). The decrease in large debris in The Gully observed in this study likely reflects the decline in fishing effort due to the collapse of the cod fishery starting in the early 1990s (Myers et al., 1997; Sinclair et al., 2015), and no-take exclusions within Zone 1 of the MPA established in 2004. However, additional efforts to mitigate and retrieve abandoned, lost or otherwise discarded fishing gear in the region may have also contributed to help reduce the input of large debris. Several fishing and non-government organizations have participated in this work, while Fisheries and Oceans Canada has initiated programs that more closely track fishing gear over its life cycle (DFO, 2022b). Once microplastics are in the marine environment, mitigation is nearly impossible; consequently, it is the responsibility of multiple different sectors and regulators to prevent debris and pollution from entering the marine environment in the first place. A multi-pronged approach (e.g., Konecny et al., 2018; Madricardo et al., 2020) will be necessary to prevent marine debris and address the health and welfare impacts of plastics on the species the MPA was originally created to protect.

CRediT author contributions

Noreen E. Kelly: Methodology; Formal analysis; Data Curation; Visualization; Writing – Original Draft; Writing – Review & Editing; Project administration; Supervision; Funding acquisition. **Laura Feyrer:** Conceptualization; Methodology; Investigation; Data Curation; Writing – Original Draft; Visualization; Writing – Review & Editing; Project administration; Supervision; Funding acquisition. **Heidi Gavel:** Methodology; Investigation; Writing – Review & Editing. **Olga Trela:** Methodology; Investigation; Writing – Review & Editing. **Wayne Ledwell:** Methodology; Investigation; Data Curation; Funding acquisition; Resources. **Heather Breeze:** Writing – Review & Editing; Funding acquisition; Project administration. **Emmaline C. Marotte:** Investigation; Writing – Review & Editing. **Leah McConney:** Investigation; Writing – Review & Editing. **Hal Whitehead:** Conceptualization; Methodology; Data Curation; Investigation; Writing – Review & Editing; Resources; Supervision; Funding acquisition; Project administration.

Declaration of competing interest

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

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envres.2023.115686.

References

Alexiadou, P., Foskolos, I., Frantzis, A., 2019. Ingestion of macroplastics by odontocetes of the Greek seas, eastern Mediterranean: often deadly. Mar. Pollut. Bull. 146, 67–75. https://doi.org/10.1016/j.marpolbul.2019.05.055.

Andrady, A.L., 2017. The plastic in microplastics: a review. Mar. Pollut. Bull. 119, 12–22. https://doi.org/10.1016/j.marpolbul.2017.01.082.

Athey, S.N., Erdle, L.M., 2022. Are we underestimating anthropogenic microfiber pollution? A critical review of occurrence, methods, and reporting. Environ. Toxicol. Chem. 41, 822–837. https://doi.org/10.1002/etc.5173.

Barnes, D.K.A., Galgani, F., Thompson, R.C., Barlaz, M., 2009. Accumulation and fragmentation of plastic debris in global environments. Philos. Trans. R. Soc. B Biol. Sci. 364, 1985–1998. https://doi.org/10.1098/rstb.2008.0205.

Barnes, D.K.A., Morley, S.A., Bell, J., Brewin, P., Brigden, K., Collins, M., Glass, T., Goodall-Copestake, W.P., Henry, L., Laptikhovsky, V., Piechaud, N., Richardson, A., Rose, P., Sands, C.J., Schofield, A., Shreeve, R., Small, A., Stamford, T., Taylor, B., 2018. Marine plastics threaten giant Atlantic marine protected areas. Curr. Biol. 28, R1137–R1138. https://doi.org/10.1016/j.cub.2018.08.064.

Barrows, A.P.W., Neumann, C.A., Berger, M.L., Shaw, S.D., 2017. Grab vs. neuston tow net: a microplastic sampling performance comparison and possible advances in the field. Anal. Methods 9, 1446–1453. https://doi.org/10.1039/C6AY02387H.

Battaglia, F.M., Beckingham, B.A., McFee, W.E., 2020. First report from North America of microplastics in the gastrointestinal tract of stranded bottlenose dolphins (*Tursiops truncatus*). Mar. Pollut. Bull. 160, 111677 https://doi.org/10.1016/j.marpolbul.2020.111677.

Baulch, S., Perry, C., 2014. Evaluating the impacts of marine debris on cetaceans. Mar. Pollut. Bull. 80, 210–221. https://doi.org/10.1016/j.marpolbul.2013.12.050.

Beer, S., Garm, A., Huwer, B., Dierking, J., Nielsen, T.G., 2018. No increase in marine microplastic concentration over the last three decades – a case study from the Baltic Sea. Sci. Total Environ. 621, 1272–1279. https://doi.org/10.1016/j.scitotenv.2017.10.101.

Benjaminsen, T., Christensen, I., 1979. The natural history of the bottlenose whale, *Hyperoodon ampullatus* (Forster). In: Winn, H.E., Olla, B.L. (Eds.), Behavior of Marine Animals. Springer US, Boston, MA, pp. 143–164. https://doi.org/10.1007/978-1-4684-2985-5_5.

Borrelle, S.B., Ringma, J., Law, K.L., Monnahan, C.C., Lebreton, L., McGivern, A., Murphy, E., Jambeck, J., Leonard, G.H., Hilleary, M.A., Eriksen, M., Possingham, H.P., De Frond, H., Gerber, L.R., Polidoro, B., Tahir, A., Bernard, M., Mallos, N., Barnes, M., Rochman, C.M., 2020. Predicted growth in plastic waste exceeds efforts to mitigate plastic pollution. Science 369, 1515–1518. https://doi.org/10.1126/science.aba3656.

Butler, S., Ibarra, D., Coffen-Smout, S., 2019. Maritimes region longline and trap fisheries footprint mapping for marine spatial planning and risk assessment. Can. Tech. Rep. Fish. Aquat. Sci. 3293 v + 30 pp.

Cai, H., Du, F., Li, L., Li, B., Li, J., Shi, H., 2019. A practical approach based on FT-IR spectroscopy for identification of semi-synthetic and natural celluloses in microplastic investigation. Sci. Total Environ. 669, 692–701. https://doi.org/10.1016/j.scitotenv.2019.03.124.

Carpenter, E.J., Smith, K.L., 1972. Plastics on the Sargasso sea surface. Science 175, 1240–1241. https://doi.org/10.1126/science.175.4027.1240.

Catarino, A.I., Macchia, V., Sanderson, W.G., Thompson, R.C., Henry, T.B., 2018. Low levels of microplastics (MP) in wild mussels indicate that MP ingestion by humans is minimal compared to exposure via household fibres fallout during a meal. Environ. Pollut. 237, 675–684. https://doi.org/10.1016/j.envpol.2018.02.069.

Chambault, P., Vandeperre, F., Machete, M., Lagoa, J.C., Pham, C.K., 2018. Distribution and composition of floating macro litter off the Azores archipelago and Madeira (NE Atlantic) using opportunistic surveys. Mar. Environ. Res. 141, 225–232. https://doi.org/10.1016/j.marenvres.2018.09.015.

Charlton-Howard, H.S., Bond, A.L., Rivers-Auty, J., Lavers, J.L., 2023. 'Plasticosis': characterising macro- and microplastic-associated fibrosis in seabird tissues. J. Hazard Mater. 450, 131090 https://doi.org/10.1016/j.jhazmat.2023.131090.

Cheung, P.K., Fok, L., Hung, P.L., Cheung, L.T.O., 2018. Spatio-temporal comparison of neustonic microplastic density in Hong Kong waters under the influence of the Pearl River Estuary. Sci. Total Environ. 628–629, 731–739. https://doi.org/10.1016/j.scitotenv.2018.01.338.

CNSOPB, C.-N.S.O.P.B., 2022. Sable Offshore Energy Project [WWW Document]. URL. https://www.cnsopb.ns.ca/offshore-activity/current-activity/sable-offshore, 1.12.22.

Colton, J.B., Burns, B.R., Knapp, F.D., 1974. Plastic Particles in Surface Waters of the Northwestern Atlantic: the abundance, distribution, source, and significance of various types of plastics are discussed. Science 185, 491–497. https://doi.org/10.1126/science.185.4150.491.

Courtene-Jones, W., Quinn, B., Ewins, C., Gary, S.F., Narayanaswamy, B.E., 2019. Consistent microplastic ingestion by deep-sea invertebrates over the last four decades (1976–2015), a study from the North East Atlantic. Environ. Pollut. 244, 503–512. https://doi.org/10.1016/j.envpol.2018.10.090.

Covernton, G., Collicutt, B., Gurney-Smith, H., Pearce, C., Dower, J., Ross, P., Dudas, S., 2019. Microplastics in bivalves and their habitat in relation to shellfish aquaculture proximity in coastal British Columbia, Canada. Aquac. Environ. Interact. 11, 357–374. https://doi.org/10.3354/aei00316.

Cózar, A., Echevarría, F., González-Gordillo, J.I., Irigoien, X., Úbeda, B., Hernández-León, S., Palma, Á.T., Navarro, S., García-de-Lomas, J., Ruiz, A., Fernández-de-Puelles, M.L., Duarte, C.M., 2014. Plastic debris in the open ocean. Proc. Natl. Acad. Sci. USA 111, 10239–10244. https://doi.org/10.1073/pnas.1314705111.

Day, R.H., Shaw, D.G., 1987. Patterns in the abundance of pelagic plastic and tar in the North Pacific ocean, 1976–1985. Mar. Pollut. Bull. 18, 311–316. https://doi.org/10.1016/S0025-326X(87)80017-6.

de Mendonça, S.N., MacIsaac, K., Moore, A.M., Johnson, C.L., 2021. Microplastic sampling and analysis in surface shelf waters: a pilot study on methods and distribution in the northwest Atlantic. Can. Tech. Rep. Fish. Aquat. Sci. 3442 v + 37 pp.

De Witte, B., Devriese, L., Bekaert, K., Hoffman, S., Vandermeersch, G., Cooreman, K., Robbens, J., 2014. Quality assessment of the blue mussel (*Mytilus edulis*): comparison between commercial and wild types. Mar. Pollut. Bull. 85, 146–155. https://doi.org/10.1016/j.marpolbul.2014.06.006.

DFO, 2022a. Threat assessment for northern bottlenose whales off eastern Canada, DFO Can. Sci. Advis. Sec. Sci. Advis. Rep. 32, 2022.
DFO, 2022b. Data on lost gear reporting [WWW Document]. URL. https://www.dfo-mpo.gc.ca/fisheries-peches/management-gestion/ghostgear-equipementfantome/reporting-declaration-eng.html. accessed 1.4.23.
DFO, 2016. Recovery strategy for the northern bottlenose whale (*Hyperoodon ampullatus*), Scotian Shelf population. In: Atlantic Canadian Waters [Final]., Species at Risk Act Recovery Strategy Series. Fisheries and Oceans Canada. vii +, Ottawa, p. 70.
DFO, 2005. The Scotian Shelf: an atlas of human activities. In: Breeze, H., Horsman, T. (Eds.), Fisheries and Oceans Canada. Nova Scotia, Dartmouth.
Di Beneditto, A.P.M., Oliveira, A. da S., 2019. Debris ingestion by carnivorous consumers: does the position in the water column truly matter? Mar. Pollut. Bull. 144, 134–139. https://doi.org/10.1016/j.marpolbul.2019.04.074.
Dimitrijevic, J., Kelly, N.E., Moore, A.M., Breeze, H., Ross, P.S., 2019. Best practices for the extraction and enumeration of microplastics in various marine environmental matrices. Can. Tech. Rep. Fish. Aquat. Sci. 3334 vi + 79 pp.
Dufault, S., Whitehead, H., 1994. Floating marine pollution in 'the Gully' on the continental slope, Nova Scotia, Canada. Mar. Pollut. Bull. 28, 489–493. https://doi.org/10.1016/0025-326X(94)90522-3.
Egger, M., Schilt, B., Wolter, H., Mani, T., de Vries, R., Zettler, E., Niemann, H., 2022. Pelagic distribution of plastic debris (> 500 μm) and marine organisms in the upper layer of the North Atlantic Ocean. Sci. Rep. 12, 13465 https://doi.org/10.1038/s41598-022-17742-7.
Eisfeld-Pierantonio, S.M., Pierantonio, N., Simmonds, M.P., 2022. The impact of marine debris on cetaceans with consideration of plastics generated by the COVID-19 pandemic. Environ. Pollut. 300, 118967 https://doi.org/10.1016/j.envpol.2022.118967.
Fernández, R., Pierce, G.J., MacLeod, C.D., Brownlow, A., Reid, R.J., Rogan, E., Addink, M., Deaville, R., Jepson, P.D., Santos, M.B., 2014. Strandings of northern bottlenose whales, *Hyperoodon ampullatus* , in the north-east Atlantic: seasonality and diet. J. Mar. Biol. Assoc. U. K. 94, 1109–1116. https://doi.org/10.1017/S002531541300180X.
Feyrer, L.J., Stewart, M., Yeung, J., Soulier, C., Whitehead, H., 2021. Origin and persistence of markings in a long-term photo-identification dataset reveal the threat of entanglement for endangered northern bottlenose whales (*Hyperoodon ampullatus*). Front. Mar. Sci. 8, 620804 https://doi.org/10.3389/fmars.2021.620804.
Fossi, M.C., Baini, M., Panti, C., Baulch, S., 2018. Impacts of marine litter on cetaceans. In: Marine Mammal Ecotoxicology. Elsevier, pp. 147–184. https://doi.org/10.1016/B978-0-12-812144-3.00006-1.
Galgani, F., Brien, A.S., Weis, J., Ioakeimidis, C., Schuyler, Q., Makarenko, I., Griffiths, H., Bondareff, J., Vethaak, D., Deidun, A., Sobral, P., Topouzelis, K., Vlahos, P., Lana, F., Hassellov, M., Gerigny, O., Arsonina, B., Ambulkar, A., Azzaro, M., Bebianno, M.J., 2021. Are litter, plastic and microplastic quantities increasing in the ocean? Microplastics Nanoplastics 1, 2. https://doi.org/10.1186/s43591-020-00002-8.
Galloway, T.S., Cole, M., Lewis, C., 2017. Interactions of microplastic debris throughout the marine ecosystem. Nat. Ecol. Evol. 1, 0116 https://doi.org/10.1038/s41559-017-0116.
Gewert, B., Plassmann, M.M., MacLeod, M., 2015. Pathways for degradation of plastic polymers floating in the marine environment. Environ. Sci. Process. Impacts 17, 1513–1521. https://doi.org/10.1039/C5EM00207A.
Geyer, R., Jambeck, J.R., Law, K.L., 2017. Production, use, and fate of all plastics ever made. Sci. Adv. 3, e1700782 https://doi.org/10.1126/sciadv.1700782.
Green, D.S., Kregting, L., Boots, B., Blockley, D.J., Brickle, P., da Costa, M., Crowley, Q., 2018. A comparison of sampling methods for seawater microplastics and a first report of the microplastic litter in coastal waters of Ascension and Falkland Islands. Mar. Pollut. Bull. 137, 695–701. https://doi.org/10.1016/j.marpolbul.2018.11.004.
Greenan, B.J.W., Petrie, B.D., Cardoso, D.A., 2014. Mean circulation and high-frequency flow amplification in the Sable Gully. Deep Sea Res. Part II Top. Stud. Oceanogr. 104, 20–34. https://doi.org/10.1016/j.dsr2.2013.07.011.
Haarr, M.L., Falk-Andersson, J., Fabres, J., 2022. Global marine litter research 2015–2020: geographical and methodological trends. Sci. Total Environ. 820, 153162 https://doi.org/10.1016/j.scitotenv.2022.153162.
Halbach, M., Vogel, M., Tammen, J.K., Rüdel, H., Koschorreck, J., Scholz-Böttcher, B.M., 2022. 30 years trends of microplastic pollution: mass-quantitative analysis of archived mussel samples from the North and Baltic Seas. Sci. Total Environ. 826, 154179 https://doi.org/10.1016/j.scitotenv.2022.154179.
Han, G., 2003. Three-dimensional seasonal-mean circulation and hydrography on the eastern Scotian Shelf. J. Geophys. Res. 108, 3136. https://doi.org/10.1029/2002JC001463.
Hartmann, N.B., Hüffer, T., Thompson, R.C., Hassellöv, M., Verschoor, A., Daugaard, A. E., Rist, S., Karlsson, T., Brennholt, N., Cole, M., Herrling, M.P., Hess, M.C., Ivleva, N. P., Lusher, A.L., Wagner, M., 2019. Are we speaking the same language? Recommendations for a definition and categorization framework for plastic debris. Environ. Sci. Technol. 53, 1039–1047. https://doi.org/10.1021/acs.est.8b05297.
Hermabessiere, L., Dehaut, A., Paul-Pont, I., Lacroix, C., Jezequel, R., Soudant, P., Duflos, G., 2017. Occurrence and effects of plastic additives on marine environments and organisms: a review. Chemosphere 182, 781–793. https://doi.org/10.1016/j.chemosphere.2017.05.096.
Hidalgo-Ruz, V., Gutow, L., Thompson, R.C., Thiel, M., 2012. Microplastics in the marine environment: a review of the methods used for identification and quantification. Environ. Sci. Technol. 46, 3060–3075. https://doi.org/10.1021/es2031505.
Hooker, S.K., De Soto, N.A., Baird, R.W., Carroll, E.L., Claridge, D., Feyrer, L., Miller, P.J. O., Onoufriou, A., Schorr, G., Siegal, E., Whitehead, H., 2019. Future directions in research on beaked whales. Front. Mar. Sci. 5, 514. https://doi.org/10.3389/fmars.2018.00514.
Hooker, S.K., Iverson, S.J., Ostrom, P., Smith, S.C., 2001. Diet of northern bottlenose whales inferred from fatty-acid and stable-isotope analyses of biopsy samples. Can. J. Zool. 79, 1442–1454.
Huntington, A., Corcoran, P.L., Jantunen, L., Thaysen, C., Bernstein, S., Stern, G.A., Rochman, C.M., 2020. A first assessment of microplastics and other anthropogenic particles in Hudson Bay and the surrounding eastern Canadian Arctic waters of Nunavut. FACETS 5, 432–454. https://doi.org/10.1139/facets-2019-0042.
Jambeck, J.R., Geyer, R., Wilcox, C., Siegler, T.R., Perryman, M., Andrady, A., Narayan, R., Law, K.L., 2015. Plastic waste inputs from land into the ocean. Science 347, 768–771. https://doi.org/10.1126/science.1260352.
Kassambara, A., 2021. Rstatix: pipe-friendly framework for basic statistical tests. R package version 0.7.0. https://CRAN.R-project.org/package=rstatix.
Kenchington, T.J., Cochrane, N.A., Gjerdrum, C., Greenan, B.J.W., Lirette, C., Moors-Murphy, H., Thompson, S.E., 2014. The environmental background to the 2008, 2009 and 2010 meso- and bathypelagic trawl surveys of the Gully. Can. Tech. Rep. Fish. Aquat. Sci. 3114, vi+251p.
Kenchington, T.J., Themelis, D.E., DeVaney, S.C., Kenchington, E.L., 2020. The meso- and bathypelagic fishes in a large submarine canyon: assemblage structure of the principal species in the Gully marine protected area. Front. Mar. Sci. 7, 181. https://doi.org/10.3389/fmars.2020.00181.
Konecny, C., Fladmark, V., De la Puente, S., 2018. Towards cleaner shores: assessing the Great Canadian Shoreline Cleanup's most recent data on volunteer engagement and litter removal along the coast of British Columbia, Canada. Mar. Pollut. Bull. 135, 411–417. https://doi.org/10.1016/j.marpolbul.2018.07.036.
Kooi, M., Nes, E.H. van, Scheffer, M., Koelmans, A.A., 2017. Ups and downs in the ocean: effects of biofouling on vertical transport of microplastics. Environ. Sci. Technol. 51, 7963–7971. https://doi.org/10.1021/acs.est.6b04702.
Kooi, M., Reisser, J., Slat, B., Ferrari, F.F., Schmid, M.S., Cunsolo, S., Brambini, R., Noble, K., Sirks, L.-A., Linders, T.E.W., Schoeneich-Argent, R.I., Koelmans, A.A., 2016. The effect of particle properties on the depth profile of buoyant plastics in the ocean. Sci. Rep. 6, 33882 https://doi.org/10.1038/srep33882.
Kühn, S., van Franeker, J.A., 2020. Quantitative overview of marine debris ingested by marine megafauna. Mar. Pollut. Bull. 151, 110858 https://doi.org/10.1016/j.marpolbul.2019.110858.
Law, K.L., Morét-Ferguson, S., Maximenko, N.A., Proskurowski, G., Peacock, E.E., Hafner, J., Reddy, C.M., 2010. Plastic accumulation in the North Atlantic subtropical Gyre. Science 329, 1185–1188. https://doi.org/10.1126/science.1192321.
Lebreton, L., Andrady, A., 2019. Future scenarios of global plastic waste generation and disposal. Palgrave Commun. 5, 6. https://doi.org/10.1057/s41599-018-0212-7.
Lebreton, L., Royer, S.-J., Peytavin, A., Strietman, W.J., Smeding-Zuurendonk, I., Egger, M., 2022. Industrialised fishing nations largely contribute to floating plastic pollution in the North Pacific subtropical gyre. Sci. Rep. 12, 12666 https://doi.org/10.1038/s41598-022-16529-0.
Ledwell, W., Huntington, J., Landry, C., Ledwell, N., Hanlon, J., 2021. Entanglements in Fishing and Marine Equipment and Strandings Reported to the Whale Release and Strandings Group in Newfoundland and Labrador and a Summary of the Whale Release and Strandings Program during 2021. https://doi.org/10.5281/zenodo.5914689.
Ledwell, W., Huntington, J., Sacrey, E., Landry, C., 2020. Entanglements in Fishing Gear and Strandings Reported to the Whale Release and Strandings Group in Newfoundland and Labrador and a Summary of the Whale Release and Strandings Program during 2019. https://doi.org/10.5281/zenodo.5911318.
Lucas, Z., 1992. Monitoring persistent litter in the marine environment on Sable Island, Nova Scotia. Mar. Pollut. Bull. 24, 192–199. https://doi.org/10.1016/0025-326X(92)90529-F.
Lusher, A.L., Hernandez-Milian, G., Berrow, S., Rogan, E., O'Connor, I., 2018. Incidence of marine debris in cetaceans stranded and bycaught in Ireland: recent findings and a review of historical knowledge. Environ. Pollut. 232, 467–476. https://doi.org/10.1016/j.envpol.2017.09.070.
Madricardo, F., Ghezzo, M., Nesto, N., Mc Kiver, W.J., Faussone, G.C., Fiorin, R., Riccato, F., Mackelworth, P.C., Basta, J., De Pascalis, F., Kruss, A., Petrizzo, A., Moschino, V., 2020. How to deal with seafloor marine litter: an overview of the state-of-the-art and future perspectives. Front. Mar. Sci. 7, 505134 https://doi.org/10.3389/fmars.2020.505134.
McAlpine, D.F., Wimmer, T., Ledwell, W., Daoust, P.-Y., Bourque, L., Lawson, J.W., Bachara, W., Lucas, Z.N., Reid, G.A., Lair, S., Francis, A., Michaud, R., 2023. A review of beaked whale (Ziphiidae) stranding incidents in eastern Canada. Can. Field Nat. 136 (in press).
Moors-Murphy, H.B., 2014. Submarine canyons as important habitat for cetaceans, with special reference to the Gully: a review. Deep Sea Res. Part II Top. Stud. Oceanogr. 104, 6–19. https://doi.org/10.1016/j.dsr2.2013.12.016.
Myers, R.A., Hutchings, J.A., Barrowman, N.J., 1997. Why do fish stocks collapse? The example of cod in Atlantic Canada. Ecol. Appl. 7, 91–106. https://doi.org/10.1890/1051-0761(1997)007[0091:WDFSCT]2.0.CO;2.
Niu, H., Lee, K., Robinson, B., Cobanli, S., Li, P., 2016. Monitoring and modeling the dispersion of produced water on the Scotian Shelf. Environ. Syst. Res. 5, 19. https://doi.org/10.1186/s40068-016-0070-5.
O'Brien, K., Whitehead, H., 2013. Population analysis of Endangered northern bottlenose whales on the Scotian Shelf seven years after the establishment of a Marine Protected Area. Endanger. Species Res. 21, 273–284. https://doi.org/10.3354/esr00533.
Ostle, C., Thompson, R.C., Broughton, D., Gregory, L., Wootton, M., Johns, D.G., 2019. The rise in ocean plastics evidenced from a 60-year time series. Nat. Commun. 10, 1622. https://doi.org/10.1038/s41467-019-09506-1.

Parga Martínez, K.B., Tekman, M.B., Bergmann, M., 2020. Temporal trends in marine litter at three stations of the HAUSGARTEN observatory in the Arctic deep sea. Front. Mar. Sci. 7, 321. https://doi.org/10.3389/fmars.2020.00321.
R Core Team, 2022. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/.
Rasband, W.S., 2018. ImageJ. U. S. National Institutes of Health, Bethesda, Maryland, USA. https://imagej.nih.gov/ij/.
Ribeiro, F., O'Brien, J.W., Galloway, T., Thomas, K.V., 2019. Accumulation and fate of nano- and micro-plastics and associated contaminants in organisms. TrAC, Trends Anal. Chem. 111, 139–147. https://doi.org/10.1016/j.trac.2018.12.010.
Rivers, M.L., Gwinnett, C., Woodall, L.C., 2019. Quantification is more than counting: actions required to accurately quantify and report isolated marine microplastics. Mar. Pollut. Bull. 139, 100–104. https://doi.org/10.1016/j.marpolbul.2018.12.024.
Rochman, C.M., Brookson, C., Bikker, J., Djuric, N., Earn, A., Bucci, K., Athey, S., Huntington, A., McIlwraith, H., Munno, K., De Frond, H., Kolomijeca, A., Erdle, L., Grbic, J., Bayoumi, M., Borrelle, S.B., Wu, T., Santoro, S., Werbowski, L.M., Zhu, X., Giles, R.K., Hamilton, B.M., Thaysen, C., Kaura, A., Klasios, N., Ead, L., Kim, J., Sherlock, C., Ho, A., Hung, C., 2019. Rethinking microplastics as a diverse contaminant suite. Environ. Toxicol. Chem. 38, 703–711. https://doi.org/10.1002/etc.4371.
Rozalska, K., Coffen-Smout, S., 2020. Maritimes Region fisheries atlas: catch weight landings mapping (2014-2018) on a hexagon grid. Can. Tech. Rep. Fish. Aquat. Sci. 3373 vi + 68 pp.
Sá, S., Bastos-Santos, J., Araújo, H., Ferreira, M., Duro, V., Alves, F., Panta-Ferreira, B., Nicolau, L., Eira, C., Vingada, J., 2016. Spatial distribution of floating marine debris in offshore continental Portuguese waters. Mar. Pollut. Bull. 104, 269–278. https://doi.org/10.1016/j.marpolbul.2016.01.011.
Sá, S., Bastos-Santos, J., Araújo, H., Pereira, A.T., Ferreira, M., Sarmento, P., Vingada, J., Eira, C., 2021. Floating marine litter and their risks to cetaceans off Portugal. Mar. Pollut. Bull. 170, 112603 https://doi.org/10.1016/j.marpolbul.2021.112603.
Sameoto, D.D., Jaroszynski, L.O., 1969. Otter surface sampler: a new neuston net. J. Fish. Res. Board Can. 26, 2240–2244.
Schmid, C., Cozzarini, L., Zambello, E., 2021. Microplastic's story. Mar. Pollut. Bull. 162, 111820 https://doi.org/10.1016/j.marpolbul.2020.111820.
Senko, J., Nelms, S., Reavis, J., Witherington, B., Godley, B., Wallace, B., 2020. Understanding individual and population-level effects of plastic pollution on marine megafauna. Endanger. Species Res. 43, 234–252. https://doi.org/10.3354/esr01064.
Shan, S., Sheng, J., Greenan, B.J.W., 2014a. Physical processes affecting circulation and hydrography in the Sable Gully of Nova Scotia. Deep Sea Res. Part II Top. Stud. Oceanogr. 104, 35–50. https://doi.org/10.1016/j.dsr2.2013.06.019.
Shan, S., Sheng, J., Greenan, B.J.W., 2014b. Modelling study of three-dimensional circulation and particle movement over the Sable Gully of Nova Scotia. Ocean Dynam. 64, 117–142. https://doi.org/10.1007/s10236-013-0672-7.
Sinclair, M., Power, M., Head, E., Li, W.K.W., McMahon, M., Mohn, R., O'Boyle, R., Swain, D., Tremblay, J., 2015. Eastern Scotian Shelf trophic dynamics: a review of the evidence for diverse hypotheses. Prog. Oceanogr. 138, 305–321. https://doi.org/10.1016/j.pocean.2015.09.005.
Smith, A., Liboiron, M., Charron, L., McIntyre, J., Hawkins, K., McLean, K., Peddle, S., Moore, G., Walzak, M.J., Goodman, A., Fulton, L., Fredericks, S., Nodding, B., 2022. Quantification and characterization of plastics in near-shore surface waters of Atlantic Canada. Mar. Pollut. Bull. 181, 113869 https://doi.org/10.1016/j.marpolbul.2022.113869.
Soto-Navarro, J., Jordá, G., Compa, M., Alomar, C., Fossi, M.C., Deudero, S., 2021. Impact of the marine litter pollution on the Mediterranean biodiversity: a risk assessment study with focus on the marine protected areas. Mar. Pollut. Bull. 165, 112169 https://doi.org/10.1016/j.marpolbul.2021.112169.
Stanistreet, J.E., Feyrer, L.J., Moors-Murphy, H.B., 2021. Distribution, movements, and habitat use of northern bottlenose whales (*Hyperoodon ampullatus*) on the Scotian Shelf. DFO Can. Sci. Advis. Sec. Res. 34, 2021/074. vi.
ter Halle, A., Ladirat, L., Gendre, X., Goudouneche, D., Pusineri, C., Routaboul, C., Tenailleau, C., Duployer, B., Perez, E., 2016. Understanding the fragmentation pattern of marine plastic debris. Environ. Sci. Technol. 50, 5668–5675. https://doi.org/10.1021/acs.est.6b00594.
Teuten, E.L., Saquing, J.M., Knappe, D.R.U., Barlaz, M.A., Jonsson, S., Björn, A., Rowland, S.J., Thompson, R.C., Galloway, T.S., Yamashita, R., Ochi, D., Watanuki, Y., Moore, C., Viet, P.H., Tana, T.S., Prudente, M., Boonyatumanond, R., Zakaria, M.P., Akkhavong, K., Ogata, Y., Hirai, H., Iwasa, S., Mizukawa, K., Hagino, Y., Imamura, A., Saha, M., Takada, H., 2009. Transport and release of chemicals from plastics to the environment and to wildlife. Philos. Trans. R. Soc. B Biol. Sci. 364, 2027–2045. https://doi.org/10.1098/rstb.2008.0284.
Van Cauwenberghe, L., Devriese, L., Galgani, F., Robbens, J., Janssen, C.R., 2015. Microplastics in sediments: a review of techniques, occurrence and effects. Mar. Environ. Res. 111, 5–17. https://doi.org/10.1016/j.marenvres.2015.06.007.
van der Hal, N., Ariel, A., Angel, D.L., 2017. Exceptionally high abundances of microplastics in the oligotrophic Israeli Mediterranean coastal waters. Mar. Pollut. Bull. 116, 151–155. https://doi.org/10.1016/j.marpolbul.2016.12.052.
Venrick, E.L., Backman, T.W., Bartram, W.C., Platt, C.J., Thornhill, M.S., Yates, R.E., 1973. Man-made objects on the surface of the central North Pacific ocean. Nature 241, 271.
Waddell, E.N., Lascelles, N., Conkle, J.L., 2020. Microplastic contamination in Corpus Christi Bay blue crabs, *Callinectes sapidus*. Limnol. Oceanogr. Lett. 5, 92–102. https://doi.org/10.1002/lol2.10142.
Waluda, C.M., Staniland, I.J., Dunn, M.J., Thorpe, S.E., Grilly, E., Whitelaw, M., Hughes, K.A., 2020. Thirty years of marine debris in the Southern Ocean: annual surveys of two island shores in the Scotia Sea. Environ. Int. 136, 105460 https://doi.org/10.1016/j.envint.2020.105460.
Weinstein, J.E., Crocker, B.K., Gray, A.D., 2016. From macroplastic to microplastic: degradation of high-density polyethylene, polypropylene, and polystyrene in a salt marsh habitat: degradation of plastic in a salt marsh habitat. Environ. Toxicol. Chem. 35, 1632–1640. https://doi.org/10.1002/etc.3432.
Werth, A.J., 2006. Mandibular and dental variation and the evolution of suction feeding in Odontoceti. J. Mammal. 87, 579–588.
Whitehead, H., Hooker, S., 2012. Uncertain status of the northern bottlenose whale *Hyperoodon ampullatus*: population fragmentation, legacy of whaling and current threats. Endanger. Species Res. 19, 47–61. https://doi.org/10.3354/esr00458.
Woodall, L.C., Gwinnett, C., Packer, M., Thompson, R.C., Robinson, L.F., Paterson, G.L.J., 2015. Using a forensic science approach to minimize environmental contamination and to identify microfibres in marine sediments. Mar. Pollut. Bull. 95, 40–46. https://doi.org/10.1016/j.marpolbul.2015.04.044.
Wright, S.L., Thompson, R.C., Galloway, T.S., 2013. The physical impacts of microplastics on marine organisms: a review. Environ. Pollut. 178, 483–492. https://doi.org/10.1016/j.envpol.2013.02.031.
Zantis, L.J., Carroll, E.L., Nelms, S.E., Bosker, T., 2021. Marine mammals and microplastics: a systematic review and call for standardisation. Environ. Pollut. 269, 116142 https://doi.org/10.1016/j.envpol.2020.116142.