Characterizing Microplastic Pollution in Aquatic Sediments from Southwestern British Columbia, Canada

by

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Abstract

Microplastics are an emerging environmental pollutant of concern because of their potential effects on biota, their ubiquity, and their persistence in the environment. Characterizing their occurrence, composition, and spatial and temporal trends in different aquatic sedimentary environments will provide insight to the likely sources of microplastics and help diagnose relationships with sedimentary processes. Our study analyzed microplastics and sedimentary properties from 6 cores from different sedimentary environments in southwestern British Columbia (BC), spanning urban and remote locations. These sites included estuarine environments from the protected area of Clayoquot Sound, lacustrine sediments in Orchid Lake from the protected Metro Vancouver watershed, and estuarine sediments from urban Boundary Bay. We also examined temporal changes in microplastic accumulation rates in Clayoquot Sound using ^{210}Pb dated sediment cores. We detected microplastics within all sediment cores, supporting previous research suggesting the ubiquity of microplastics in aquatic sedimentary environments. We examined the sediments for all forms of microplastics and found only microfibers that were predominantly black and blue. After applying corrections for contamination, we calculated concentrations and accumulation rates for the surface 10 cm in each core (the depth over which microplastics were found). Mean (+/- SE) microplastic concentrations (#particles/kg) in Clayoquot Sound were 288 $\pm$ 92 particles/kg, approximately 2.5x greater than in Boundary Bay (114 $\pm$ 61 particles/kg) and 1.3x greater than Orchid Lake (223 $\pm$ 188 particles/kg). When compared with 42 sites from marine sedimentary environments across the globe, we found microplastic concentrations at our sites were of a similar order of magnitude to 79% of surveyed sites with mean/median microplastic concentrations ranging from 0-500 particles/kg. Microplastic accumulation rates in Clayoquot Sound increased drastically with mean (+/- SE) concentrations of 7 $\pm$ 3 particles/100cm^2/year in 1950 compared to 33 $\pm$ 12 particles/100cm^2/year in 2016. While atmospheric transport is the most likely source of microplastic deposition at Orchid Lake, multiple sources are possible at the other two locations. In Clayoquot Sound, the proximity of our sites to wastewater effluent and aquaculture sites points to these activities as potential sources. At both Clayoquot Sound and Boundary Bay, tidal currents introducing marine sources of microplastics is also likely.

Keywords: Microplastic, microfiber, microplastic concentration, microplastic accumulation rates

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Chapter 1. General Introduction

Global plastic production has increased exponentially since the early 20th century, and substantial amounts of plastic end up in the marine environment (Avio et al., 2017; Bergmann et al., 2015; Geyer et al., 2017; Lebreton & Andrady, 2019). Today, over 300 million tonnes of plastic are produced annually, and roughly 8 million tonnes end up in the world's oceans every year, contributing to 80% of marine debris (Plastics Europe, 2012). The majority of plastics reach the marine environment from a combination of land-based sources via streams, rivers, stormwater runoff and wastewater effluent, and direct marine sources via illegal or accidental dumping, with ocean basins acting as the ultimate sink for plastics (Ajith et al., 2020; Chen et al., 2018; Lusher et al., 2017; Nel & Froneman, 2015; Zhang, 2017).

Recent attention has focused on characterizing and quantifying microplastics (MPs) in the environment (Akdogan & Guven, 2019; Peng et al., 2017; Yu et al., 2020; Zhang et al., 2019). MPs are plastic particles that are less than 5mm in size and exist as either fibers, filaments, granules, pellets, or beads (Hidalgo-Ruz et al., 2013; Wu et al., 2020). MPs are characterized as primary or secondary; primary MPs are introduced to the environment in their original form, and secondary MPs form from the degradation of larger plastic products (Avio et al., 2017; Hidalgo-Ruz et al., 2013). While some MPs enter the marine environment as primary MPs, the majority of MPs are formed *in-situ* in the marine environment as they undergo physical weathering and photochemical degradation in the surface ocean (Bergmann et al., 2015).

MPs are a cause of concern because of their potentially harmful impacts to biota, their ubiquity, and their persistence in the environment (Avio et al., 2017; Franzellitti et al., 2019; Ngo et al., 2019). MPs have been shown to cause physical harm through ingestion in many aquatic organisms and pose a risk of chemical contamination from leaching of harmful plastic additives (Chen et al., 2019; Covernton et al., 2019; Desforges et al., 2015; Franzellitti et al., 2019;

Hahladakis et al., 2018; Hall et al., 2015; Luo et al., 2019; Wu et al., 2020). In addition to leaching, persistent organic pollutants and metals can adsorb to plastic particles and have the potential to bioaccumulate in the tissues of organisms and biomagnify in the marine food web. MPs are ubiquitous in that they have been detected in aquatic sediments around the world, from remote locations in the deep-sea, Arctic Ocean, and alpine lakes, to populated coastal regions (Bergmann et al., 2017; Brandon et al., 2019; Free et al., 2014; Mohamed Nor & Obbard, 2014; Vaughan et al., 2017). MPs are also extremely persistent as they do not biodegrade in marine or terrestrial environments but rather break down into smaller fragments known as nanoplastics (Andrady, 2011). Recognizing these concerns, the European Union recently called for characterization of the amount, distribution, and composition of MPs in their Marine Strategy Framework Directive (Marine Strategy Framework Directive, Annex III, 2017).

For this project, we studied MPs in southwestern British Columbia (BC) from different aquatic sedimentary environments. Our study locations included three sites in Clayoquot Sound on the West Coast of Vancouver Island, two locations in Boundary Bay in Metro Vancouver, and one location at Orchid Lake in the Seymour Watershed. The coastal study sites were selected to represent a range of coastal sedimentary environments to test the influence of sedimentary environment on MP deposition. We sampled Orchid Lake to serve as a baseline for contamination in the region as Orchid Lake presents a closed system with effectively no anthropogenic activity. In the coastal environments of Clayoquot Sound and Boundary Bay, we postulated that both land- and water- based sources were major contributors to MP deposition at the study sites. In Clayoquot Sound, MPs from aquaculture and wastewater effluent transported in surface currents are likely sources of MP deposition because proximity to these sources has been suggested to influence MP deposition in other regions (Chen et al., 2018; Krüger et al., 2020; Ziajahromi et al., 2016). In Boundary Bay, water contaminated with MPs from the Strait of Georgia and Puget Sound is likely a major source of MPs to the sites (Johannessen & Ross, 2002; Johannessen et

al., 2015; Macdonald et al., 1991; Yunker et al., 1999). Orchid Lake, in contrast, is located in the nearby Seymour Watershed, with no sources of riverine inflow from nearby human settlements or activities. Similar to sites from other remote alpine and subalpine lakes (Allen et al., 2019; Feng et al., 2020), MPs are likely introduced to this lake through atmospheric deposition.

Studies on MPs in coastal BC are limited, and those that exist have focused on MPs in the water column and in marine species (Cluzard et al., 2015; Collicutt et al., 2019; Desforges et al., 2014). For this study, we measured MP concentrations in sediment cores from each study location. Analyzing MPs in sediment cores allowed us to study the depositional trends of MPs at depth. In the sediments of Clayoquot Sound, ^{210}Pb dated cores from a previous study (Postlethwaite et al., 2018) were used to provide chronological control to estimate MP deposition rates, making our study the first to analyze temporal changes in MPs at depth in aquatic sediments in BC. Establishing rates of change in MP deposition allowed us to determine a depth range or period of time over which MPs have been accumulating in the sediments and to use this time range to draw comparisons to MP deposition at other sites. Understanding changes in microplastic accumulation rates over time was also useful for identifying potential sources of MPs to the study sites.

Factors controlling the deposition and distribution of MPs in ocean and sedimentary environments are complex, and many processes can operate at once. Once in the marine environment, plastics are subject to oceanographic transport via surface and bottom water currents, thermohaline circulation, and tidal processes (Alimba & Faggio, 2019; Gago et al., 2017; Loughlin et al., 2021; Zhang, 2017). Polymers that are denser than seawater sink into the sediment (Alimba & Faggio, 2019; Gago et al., 2017; Loughlin et al., 2021; Zhang, 2017). Lower density plastic polymers can also make their way into the sediment through biofouling, marine snow, fecal pellet aggregation, and downwelling (Loughlin et al., 2021). Once in the sediment, several factors influence MP distribution. Bioturbation from burrowing organisms can rework and displace MPs downward into the sediments, and upwelling and high energy events can resuspend MPs into sea water (Harris,

2020; Martin et al., 2017; Zhang, 2017). In this study, we considered which oceanographic processes could influence MP deposition at the marine sites and also studied the influence of sedimentary characteristics such as organic carbon content ($\%C_{org}$) and sediment grain size distribution. While the literature is inconclusive (Dodson et al., 2020; Mathalon & Hill, 2014), some studies have found that MPs settle in lower energy environments associated with a higher percentage of fine sediments and higher organic carbon content (Courtene-Jones et al., 2020; Maes et al., 2017; Vianello et al., 2013).

In Chapter 2 we analyzed MPs in dated sediment cores from the tidal flats of Clayoquot Sound to explore changes in MP deposition rates through time and to better understand the relationship between MP deposition and sedimentary environment. We also compared MP concentrations and accumulation rates in Clayoquot Sound to values documented in other marine sedimentary environments from around the globe to study the influence of marine sedimentary environment and location on MP deposition in marine sediments, and to place our study sites within the context of the broader literature. Chapter 2 is formatted for publication in the journal PlosONE and answers the questions:

- How have the accumulation rates of MPs changed through time in Clayoquot Sound?
- Does the sedimentary environment influence the deposition of MPs?
- How do MP concentrations and accumulation rates at our sites compare to sites from different marine sedimentary environments around the globe, and what inferences can we make from these comparisons?
- What are the likely sources of MPs to the region?

In Chapter 3 we described the occurrence, trends, composition, and spatial distribution of MPs in a remote mountain catchment, Orchid Lake, as well as an urban bay in Metro Vancouver, Boundary Bay. Chapter 3 builds on Chapter 2 by further illuminating the potential controls on MP

deposition in urban and remote environments. This work represents work that was conducted in collaboration with Parks Canada and Metro Vancouver and answers the questions:

- How does MP deposition in the sediments of a closed system in a sub-alpine lake compare to marine coastal sediments of the region?
- Does location, urban vs. remote, impact MP deposition?
- What are the likely sources of MPs to the sites?
- Does sedimentary environment impact MP deposition?

In the concluding chapter we compared MP deposition at the various study locations and discussed what our research tells us about the likely sources of MPs to the region, as well as the major factors that control MP deposition at the study sites. We also discussed the start of the Anthropocene and the role of MPs as a sedimentary indicator, the major gaps in our research, next steps in MP research in aquatic sedimentary environments, and some of the management implications of our findings.

Chapter 2. Temporal Trends in Microplastic Deposition in Marine Sedimentary Environments in Clayoquot Sound, British Columbia, Canada

2.1. Abstract

Understanding temporal trends in microplastic deposition in marine sedimentary environments can help to determine relationships between microplastics and sedimentary processes, as well as potential sources. Here we measured microplastic concentrations in three ^{210}Pb dated sediment cores to characterize temporal changes in microplastic deposition in sediments along the Pacific Coast of Canada, within the Clayoquot Sound UNESCO Biosphere Reserve and the Pacific Rim National Park Reserve. Sediment grain size distribution and percent organic carbon content ($\%C_{org}$) of the sediments were measured to better understand the sedimentary environment at the sites, and to examine potential relationships between sedimentary environment and microplastic deposition. While all microplastic types were considered, the microplastics discovered in this study were all microfibers, which corroborates other research in the area placing high quantities of microfibers in the subsurface seawater of the northeastern Pacific Ocean. In the surface 10 cm of sediment (where most microplastics were deposited), mean (+/- SE) microplastic concentrations (#particles/kg) in Clayoquot Sound were 288 $\pm$ 92 particles/kg, which is of a similar order of magnitude to 79% of sites studied in a metanalysis of 42 sites around the globe. Microplastic concentrations were not evenly distributed throughout the core and displayed a high degree of variability at depth. While few studies have measured microplastic accumulation rates in marine sedimentary environments, mean values measured over the last 30 years in Clayoquot Sound are on a similar order of magnitude to those documented for the same time period at all other sites that have reported microplastic accumulation rates globally, ranging from 2-35 particles/100cm^2/year. Mean (+/- SE) microplastic accumulation rates in the region increased from 7 $\pm$ 3 particles/100cm^2/year in 1950 to 33 $\pm$ 12

particles/100cm^2/year in 2016, with increases coinciding with global plastic production and increases in aquaculture tenures in Clayoquot Sound. A significant relationship was observed between microplastic concentrations and accumulation rates and the %fine sediment grain size fraction, implying that microplastics may preferentially be deposited in finer sediments associated with lower energy environments.

2.2. Introduction

Studies on microplastics (MPs) across the globe, in both the water column and in sediments, have found the highest concentrations of MPs in coastal areas (e.g. Andrady, 2011; Cluzard et al., 2015; Desforges et al., 2014). MPs can be transported across the open ocean before accumulating in coastal areas that are subject to restricted water movement (Zhang, 2017). MPs from land-based and marine sources are prevalent in coastal areas and appear in both sediments as well as the water column (Alimba & Faggio, 2019; Bergmann et al., 2015; Gies et al., 2018; Ziajahromi et al., 2017). Lebreton et al. (2019) modeled the fate of plastic debris in coastal environments and showed that plastics have a high residence time, on the order of several years to decades, in coastal areas. Based on their model, Lebreton et al. (2019) estimated that 66.8% of all buoyant plastics released into the marine environment since the 1950s remain in the coastal environment. These findings demonstrate that coastal environments are important areas for studying the depositional trends of MPs.

The marine sedimentary environment has been described as a major sink for MPs (Frias et al., 2016; Pagter et al., 2020; Peng et al., 2017), and therefore, documenting accumulation hotspots and quantifying MP concentrations on the seafloor is important for understanding implications for marine ecosystems. There are a number of means by which MPs make their way into the marine environment and ultimately settle on the seafloor. Major MP pathways include sewage systems, riverine inputs, storm water outflows, maritime activities, improper disposal, atmospheric

deposition, and *in situ* environmental formation from larger plastic products (Bergmann et al., 2015; Browne et al., 2011; Zhang, 2017). Once in the marine environment, a number of processes such a biofouling and incorporation with marine aggregates can modify the density of MPs and result in the increased density of MPs, and thus increased deposition (Loughlin et al., 2021). In the sediment itself, vertical distribution of MPs is controlled by a number of factors and processes such as trawling, bioturbation, tidal forcing, and weather events which can all redistribute MPs in the sediment column (Loughlin et al., 2021; Van Cauwenberghe et al., 2015). Understanding factors that control the distribution and accumulation of MPs in sedimentary environments could help provide insight to impacts on marine ecosystems, and in particular benthic organisms. Taylor et al., (2016) found that redistribution of MPs in sediments resulted in repeated exposure to deposit feeders in benthic environments, putting these organisms at higher risk than filter feeders in the same environments.

In this study we quantified MP concentrations and accumulation rates to provide insight to the fate of MPs in the sediments of Clayoquot Sound, British Columbia, Canada. Clayoquot Sound is located on the West Coast of Vancouver Island on the Pacific Coast of Canada within the Clayoquot Sound UNESCO Biosphere Reserve and the Pacific Rim National Park Reserve. We chose these sites to be representative of different sedimentary environments in the region, as well as for their remote location, with limited population and accessibility. Clayoquot Sound, while low in population density and remote in location, contains wastewater discharge from nearby Tofino and holds 17 active aquaculture tenures, both of which could influence MP concentrations and accumulation rates (Province of British Columbia, 2021). These factors have been shown to influence MP deposition in other locations, and are further investigated here (Blair et al., 2019; Krüger et al., 2020).

While a number of studies have focused on characterizing the variability of MPs across the globe (Harris, 2020), less attention has been paid to characterizing the vertical distribution of MPs in the

sediment column. Our study provides the first analysis of temporal changes in MP accumulation rates at depth in coastal sediments in BC. This temporal component allowed us to determine an age or depth horizon over which MPs have been accumulating in the sediments and allowed us to make inferences about potential source dynamics as well as draw better comparisons to other sites. To study how sedimentary environment might affect MP deposition within the Sound, we examined the sediment grain size distribution and percent organic carbon ($\%C_{org}$) of the sediments to test the hypothesis that MPs tend to accumulate in finer sediments with higher $\%C_{org}$ associated with lower energy environments (Courtene-Jones et al., 2020; Maes et al., 2017; Vianello et al., 2013). Finally, we placed MP concentrations and accumulation rates from Clayoquot Sound within the context of the broader literature to better understand the influence of location and environment type on MP deposition in marine sediments.

2.3. Methods

2.3.1. Study Area

Sediments were sampled from three sites in Clayoquot Sound on the west coast of Vancouver Island, BC. Clayoquot Sound is relatively remote with limited accessibility due to its location on the west coast of Vancouver Island, with a low population encompassing a vast area of 2600 km^2 (Figure 1) (Province of British Columbia, 2017). The nearest town, the District of Tofino, has a population of only 1,932 people (Statistics Canada, 2016). Port Alberni, 127km away, is the closest city and has a population of 17,678 people while the whole of Vancouver Island has a population of 870,297 (Statistics Canada, 2016).

Although the location is remote and the population density relatively low, the area is an active region for tourism and fisheries. In 2018, an estimated 600,000 visitors passed through Tofino (Tourism Tofino, 2020). Aside from tourism, fisheries is the primary industry in the region. The development of aquaculture in the region has increased drastically since the first tenure was

established during the mid 1980s (Flaherty et al., 2019; Lusher et al., 2017). Currently, 17 aquaculture tenures, representing a cumulative area of 855 hectares, are active in Clayoquot Sound (Figure 1) (Province of British Columbia, 2021). All three of the study locations are within approximately 5 km distance from active aquaculture tenures, which are possible sources of MPs.

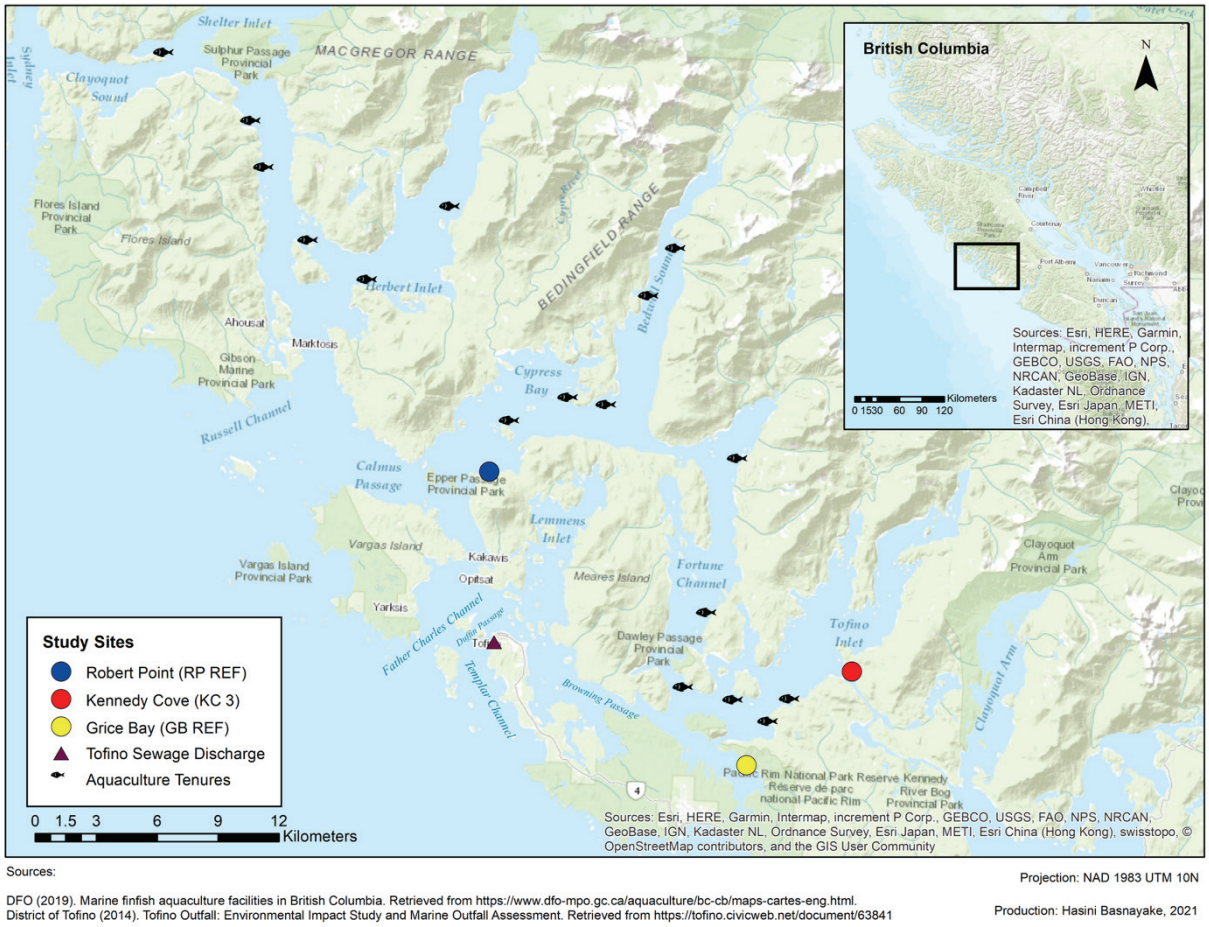


Figure 1. Map depicting the three sampling locations: Grice Bay (GB REF), Kennedy Cove (KC 3) and Robert Point (RP REF) in Clayoquot Sound, British Columbia, Canada. The triangle represents the District of Tofino's sewage outflow (District of Tofino, 2017). The fish represent active aquaculture tenures in Clayoquot Sound (Province of British Columbia, 2021).

The three sites, Robert Point, Grice Bay, and Kennedy Cove are located in the southern region of Clayoquot Sound (Figure 1). These sites were selected to provide accurate representation of intertidal sediments from the region which are exposed to a range of oceanographic conditions

and sedimentary environments that might act as controls for MP deposition. Robert Point is the most exposed site, with the greatest interaction with the open ocean, demonstrated by its higher salinity value of 29.3 ± 0.5 (Postlethwaite et al., 2018). It is located to the north of Meares Island at the confluence of the Father Charles Channel and the Calmus Passage. Grice Bay and Kennedy Cove are more sheltered sites, located deep within the Tofino Inlet which is isolated from the coastal shelf region of Clayoquot Sound by Meares Island. Its main connections to the outer, coastal shelf of Vancouver Island are through the Templar and Father Charles Channels, located to the southwest of Meares Island. Grice Bay is located in the Tofino Inlet further from the open ocean with a salinity of 24.1 ± 0.1 (Postlethwaite et al., 2018). The core at Kennedy Cove was taken close to the outflow of the Kennedy River, the dominant freshwater source draining into the Tofino Inlet. This freshwater influence is evident from its low salinity value of 6.3 ± 0.4 (Postlethwaite et al., 2018).

Both the Tofino Inlet and the Calmus Passage are fjord-type waterways with shallow sills and narrow connecting passages which restrict connection to the open sea (Coote, 1965). The shallow sills restrict incoming ocean water to the upper 11-13 meters of the surface layer (Coote, 1965). These fjord-like systems tend to have strong tidal mixing, so the MPs are likely mixed to deeper layers once they cross the sills into the Sound. Once inside Tofino Inlet, the direction and speed of the currents in the Inlet tend to follow tidal oscillations, flowing to the northeast during a flood tide and southwest during an ebb tide (Worley Parsons, 2014; Environment Canada, 2009).

Ocean currents along the west coast of Vancouver Island also influence MP transport to Clayoquot Sound and exhibit seasonal variation. In winter, the Alaskan Current moves towards the northwest, parallel to shore (Freeland et al., 1984). In summer, a southeastward current that is part of the California Current System develops near the shelf break becoming strongest in August (Freeland et al., 1984). Both of these currents carry water from the North Pacific Ocean,

where significant plastic debris as part of the Great Pacific Garbage Patch is known to exist in the subsurface. Nearshore during summer, the Vancouver Island Coastal Current (VICC) is driven by outflow from the Fraser River and flows toward the northwest along Vancouver Island's outer coast to Brooks Peninsula, (Freeland et al. 1984; Okey & Dallimore, 2015). This current carries water from the Salish Sea estuary and contaminated Fraser River to the West Coast of Vancouver Island (Freeland et al., 1984).

2.3.2. Field Methods

Three sediment cores were collected in May and June of 2016 as part of a previous study examining blue carbon storage in eelgrass meadows (Postlethwaite et al., 2018). The cores were retrieved from intertidal and subtidal zones, which were visually determined using the low tide mark (Table 1). The cores at Grice Bay and Robert Point were collected from bare sediment in the intertidal zone, referred to as reference cores. The Kennedy Cove core was collected from an eelgrass meadow further offshore in the subtidal zone. Cores were taken using a simple push method where three-inch polycarbonate tubes were beveled on one end to help cut through sediment and then pushed into the sediment until depth of refusal. This method resulted in minimal (<2 cm) compaction. Cores were extracted in the field at 1-cm intervals into sterile sample bags and were handled carefully so as not to disturb soil compaction. The bags were kept in coolers until they were brought back to the laboratory and refrigerated at 4°C.

Table 1. Information for cores collected from Grice Bay (GB REF), Kennedy Cove (KC 3) and Robert Point (RP REF).

Location	Core ID	Latitude	Longitude	Date Collected	Core length	Age at Top Relative to 2016 (years)	Age at Bottom Relative to 2016 (years)
Grice Bay	GB REF	49.099	-125.738	08-06-2016	28 cm	3.05	103.38
Kennedy Cove	KC 3	49.142	-125.669	07-06-2016	39 cm	3.00	124.90
Robert Point	RP REF	49.225	-125.919	27-05-2016	33 cm	2.90	104.10

2.3.3. Microplastic Extraction

Each 1-cm sample was thawed to room temperature and homogenized, and a weighed amount of dried material was removed from the subsample to be used for MP extraction. Density separation techniques are the most common method of MP extraction from sediments, and sodium polytungstate with a density of 1.4 g/cm^3 has been proven to be especially effective at MP extractions (Hanvey et al., 2017; Mathalon & Hill, 2014; Stolte et al., 2015; Zobkov & Esiukova, 2017). A carefully weighed amount of sediment was added to a 50 mL centrifuge tube containing 20mL of sodium polytungstate solution. The samples were shaken for 5 minutes and were then spun at 3000 rpm for 10 minutes. The supernatant was then transferred from the 50 mL centrifuge tube to a 15 mL centrifuge tube and re-spun at 3000 rpm for an additional 10 minutes to ensure all MPs were separated from the sediments (Claessens et al., 2011; Van Cauwenberghe et al.,

2015). Following the second centrifugation, the supernatant was pipetted onto Whatman50 (2.7 µm pore size) filter paper in a Büchner funnel. The Büchner funnel was quickly covered with aluminum foil to prevent MP contamination from ambient air. Filter papers were left to filter for 24 hours and were subsequently transferred to covered petri dishes for microscopic analysis.

MPs were identified under a Leica M205C stereomicroscope using the Hidalgo-Ruz et al. (2012) MP characteristics guide (Appendix A). MPs were characterized based on their small size (<5 mm), lack of cellular or organic structures, uniformity in thickness throughout their length, and clear and homogenous colour throughout (Hidalgo-Ruz et al., 2013). We were open to examining all types of MPs including fibers, fragments, granules, filaments, beads, and pellets. MPs were visually identified, counted, and categorized based on type, size, and colour (Appendix B).

2.3.4. Contamination Control

Care was taken at all stages of the study to minimize contamination of the samples from airborne MPs. In the laboratory, all instruments were rinsed with filtered deionized water between each step of the procedure. Cotton as opposed to synthetic clothing was always worn, and fans with negative pressure were run to prevent settling of airborne MPs. To quantify contamination from airborne MPs during laboratory analysis, laboratory control filter papers were placed on the work bench and in the fume hood. Additionally, procedural blanks of deionized water were run through the entire extraction process to determine the amount of contamination during analysis. The number of MPs on the laboratory control and procedural blank filters were counted and used to calculate daily contamination values (Appendix C). The daily contamination values (n=3) were then averaged to produce an average contamination value unique to each core. Average daily contamination values ranged from 1 to 4 MPs. Across all sampling days, the average MP contamination value was 3 MPs. All data presented have been adjusted to account for average daily contamination values.

2.3.5. Microplastic Concentrations

MP concentrations were calculated for each 1-cm subsection of the core (Appendix D). The numbers of MPs per gram (and per kilogram) were calculated based on the dried weight of each sample (Appendix E).

2.3.6. ^{210}Pb Dating (CRS Model)

The sediment cores were dated using previously published age models established for each core based on measurements of the naturally occurring radionuclide ^{210}Pb (Postlethwaite et al., 2018). The use of ^{210}Pb dating of sediments is widely accepted and has been used to study the relationship between anthropogenic activities and contaminants such as trace metals and organic pollutants (Kirchner, 2011). The two main models used to date sediments include the constant rate of supply (CRS) model, and the constant initial concentration (CIC) model (Turner & Delorme, 1996). While both models assume a cosntant flux of ^{210}Pb through time, the CIC model assumes that the sediment accumulation rate has remained constant over the dating interval, while the CRS model allows for fluctuations in sediment accumulataion rates over time (Abril, 2019). For the cores in Clayoquot Sound, the CRS model, which assumes sedimentation rates vary with depth, was chosen. While the CRS model has been shown to be an effective model for sediment dating, a study by Barsanti et al., (2020) on the efficacy of ^{210}Pb for sediments found that several factors including the condition of the core (disturbed vs. undisturbed), slicing thickness, and selection of the appropriate model can result in significantly diferent chronologies. One way to deal with this uncertainty in the dating models would be to use an additional marker, such as ^{137}Cs, with a spike around ~1960, to test the efficacy of the model (Barsanti et al., 2020).

The sediment ages for the top 1 cm of the cores ranged from 2.90 to 3.05 years old (relative to 2016), and the basal ages ranged from 104.10 to 124.90 years old (Table 1).

2.3.7. Microplastic Accumulation Rates

The ^{210}Pb chronologies were used to determine the sediment mass accumulation rates for each 1-cm subsection of the cores (Appendix F). First, the linear sedimentation rate (LSR, cm/y) was calculated for each 1-cm subsection of the cores using Equation 1:

$$\mathrm{LSR} = \frac{(\mathrm{N2}-\mathrm{N1})}{(\mathrm{T2}-\mathrm{T1})} \quad (1)$$

Where, N2 is the lower section depth (cm), N1 is the upper section depth (cm), T2 is the age at bottom of lower depth in years, and T1 is the age at bottom of upper depth in years.

Sediment mass accumulation rates (SMAR, g/cm^2/year) were then calculated for each 1-cm subsection of the core using Equation 2:

$$\mathrm{SMAR} = \mathrm{LSR} * \mathrm{DBD} \quad (2)$$

where DBD is dry bulk density (g/cm^3). To measure DBD, a known volume of sediment subsectioned from the core was subsampled and dried at 60°C for 72 hours. The sediment was then weighed to obtain g/cm^3

Microplastic accumulation rates (MPARs) (#particles/cm^2/year) (Appendix F) were then determined for each 1-cm subsection of the core using Equation 3:

$$\mathrm{MPAR} = \mathrm{SMAR}\ \mathrm{x}\ [\mathrm{MP}] \quad (3)$$

where [MP] is microplastic concentration (#particles/g).

2.3.8. Sediment Grain Size Analysis

Sediment grain size distribution was also completed for a subset of samples (Appendix G) to provide further information on the energetic environment at each site, with coarser sediments typically representing higher energy environments, and finer sediments indicative of lower energy environments (Enders et al., 2019). We anticipated that MPs might respond in a similar manner to suspended sediment in that they are likely to settle preferentially in less energetic environments (Enders et al., 2019; Haave et al., 2019; Zhang et al., 2019). Thus, finer sediments, which indicate lower energy environments, may be more likely to be associated with higher concentrations and deposition rates of MPs. To measure grain size, samples were dried at 60°C for 24 hours and then sorted with a series of graded sieves. A known weight of subsample was poured over a sieve stack with mesh sizes of 0.0625 mm, 0.125 mm, 0.25 mm, 0.5 mm, 1 mm and >1mm. The sieve stack was shaken for 10 minutes. The sieved sediments from each size fraction were weighed, and a dry weight percentage of each size fraction was calculated. Grain size classifications were based off the Wentworth grain size classification system: <0.062 mm (silt + clay), 0.062-0.125 mm (very fine sand), 0.125-0.250 mm (fine sand), 0.25-0.5 mm (medium sand), 0.5 -1 mm (coarse sand), and >1 mm (very coarse sand) (Wentworth, 1922).

2.3.9. Percent Loss-on-Ignition and Percent Organic Carbon

The organic carbon content of the sediments was determined to provide insight into the sedimentary environment at the sites (Appendix G). Specifically, MPs that are suspended in water may aggregate with particulate organic matter and settle out of the water column to result in increased MP deposition (Haave et al., 2019; Maes et al., 2017; Vianello et al., 2013). Organic carbon content also provides insight into the energy environment at the site, with higher percentages of organic material often associated with lower energy environments (Enders et al., 2019).

To characterize the organic carbon content of the sediments at each site, percent loss-on-ignition ($\%LOI_{550}$) was measured at each 1-cm interval of the cores. $\%LOI_{550}$ was determined at each depth by combusting samples of known weight at 550°C for 4 hours, followed by re-weighing the sample and obtaining the weight difference from combustion, using Equation 4:

$$\%\mathrm{LOI}_{550} = \frac{(\mathrm{Pre}_{550} - \mathrm{Post}_{550})}{(\mathrm{Pre}_{550})} \text{ x } 100 \quad (4)$$

Where, $\%LOI_{550}$ is weight % LOI, Pre_{550} is the weight of original dry sample before 550°C combustion (g), and $Post_{550}$ is the weight of dried sample after 550°C combustion (g).

Percent organic carbon content ($\%C_{org}$) for each 1-cm interval of the cores was estimated by using the calculated $\%LOI_{550}$ and a previously published relationship (Equation 5) between %LOI and %C (Postlethwaite et al., 2018). Equation 5:

$$\%\mathrm{C}_{\mathrm{org}} = [(0.2985) * (\%\mathrm{LOI}_{550})] - 0.2552 \quad (5)$$

2.3.10. Global Comparison

We conducted a literature review on MPs in marine sedimentary environments to compare MP concentrations and accumulation rates in Clayoquot Sound to other regions across the globe to better understand the influence of location and sedimentary environment on MP deposition. The search engines Web of Science and Google Scholar were used to search for journal articles using the key words: “microplastic*”, “sediment*”, “ocean”, “coast”, “microfibre”, and “microfiber”. The search resulted in 1,210 studies.

To draw precise comparisons with our study, we selected studies carried out on marine sediments. Specifically, we examined sedimentary environments that were classified in their publications as beaches, shallow coastal environments, estuaries, lagoons, continental shelf environments, and the deep sea. Studies on MPs in lake and river sediments, the water column,

biota, and other environmental mediums were excluded. Review papers were also not included in this analysis. After limiting the results based on these criteria, 84 studies remained.

Only studies that employed similar methodologies, namely using density separation techniques to extract MPs from sediment, were used. Studies were chosen that reported MPs in #particles/kg to facilitate comparisons with our findings. Studies in which area or volume-based measurements could not be converted to #particles/kg were excluded. Of the 8 studies that reported MPs in area or volume-based measurements, none were converted to #particles/kg because the studies did not report a density measurement (Appendix J). After applying these criteria, we were able to examine mean and/or median MP concentrations in #particles/kg from 70 sites including 16 beach, 15 shallow coastal, 16 estuarine, 3 lagoon, 15 continental shelf, and 5 deep-sea environments (Appendix I).

A number of these studies did not report contamination measures or correct their data for contamination. To draw attention to this, we filtered additionally for this criteria. Of the sites analyzed, 28 did not meet this criteria. The 42 sites that remained included 9 beach, 8 shallow coastal, 9 estuarine, 12 continental shelf, and 4 deep sea environments.

Lastly, to compare MPARs from Clayoquot Sound to the broader literature, we filtered for studies that provided chronological control of sediments. Chronological control was observed in 7 studies in marine sedimentary environments. Of these studies, 1 reported MPARs and 3 provided SMARs which were used to determine MPARs. In total, 4 studies on MPARs in marine sedimentary environments were compared to our findings in Clayoquot Sound. These included 1 continental shelf, 1 deep sea, and 2 shallow coastal environments.

The majority of these studies present mean and/or median concentrations for the top 5-10cm of sediment, and MPARs over the last ~30 years. In order to draw comparisons between these

studies and ours, we averaged MP concentrations in the top 10cm of our cores and calculated mean MPARs for the last ~30 years.

2.3.11. Statistical Analysis

MP concentrations, SMARs, MPARs, $\%C_{org}$ and sediment grain size distributions were compared for significant differences between the three study sites. All data were first tested for normality using the Shapiro-Wilk test for normality. Where data were normally distributed (SMARs), One-Way ANOVAs were conducted to test for significant differences between the sites. Where data were not normally distributed (MP concentration, MPARs, $\%C_{org}$, grain size distribution, MP size), the non-parametric Kruskal-Wallis test was used.

To test for differences in sedimentary environment influencing MP concentrations and MPARs between the sites, the Wilcoxin Signed Rank Test for paired data was used to identify any statistically significant differences in paired data between the sites. The paired data in this analysis included MP concentrations and MPARs paired with $\%C_{org}$, the %fine sediment fraction (0.125-0.25 mm) and the %coarse sediment fraction (0.5-1 mm). These data were graphed for visual representation and the Wilcoxin Signed Rank Test was used to determine if any differences between sites were statistically significant.

The significance level of all tests was set at $\alpha = 0.05$.

2.4. Results

2.4.1. Sediment Properties

Sediment grain size distributions for Robert Point, Grice Bay, and Kennedy Cove were determined for the entire length of the core. Across all sites, the sediments were comprised of 66% fine sediments (made up of silt + clay, very fine sand, and fine sand) and 34% coarse sediments

(medium sand, coarse sand, and very coarse sand). Robert Point contained the finest sediments and Kennedy Cove the coarsest sediments (Figure 2). The sediments at Grice Bay and Robert Point consisted of mainly of fine sand (30% at Grice Bay and 78% at Robert Point) and very fine sand (30% at Grice Bay and 12% at Robert Point), with a small proportion of silt and clay (6% at Grice Bay and 0.12% at Robert Point). The sediments at Kennedy Cove were much coarser, consisting mainly of medium sand (16%), coarse sand (23%) and very coarse sand (28%). Both the %fine sand and %coarse sand grain size distributions were found to be significantly different between the sites (%fine: Chi square=15.47, p=0.0004*, df=2; %coarse: Chi square=17.82, p=0.0001*, df=2).

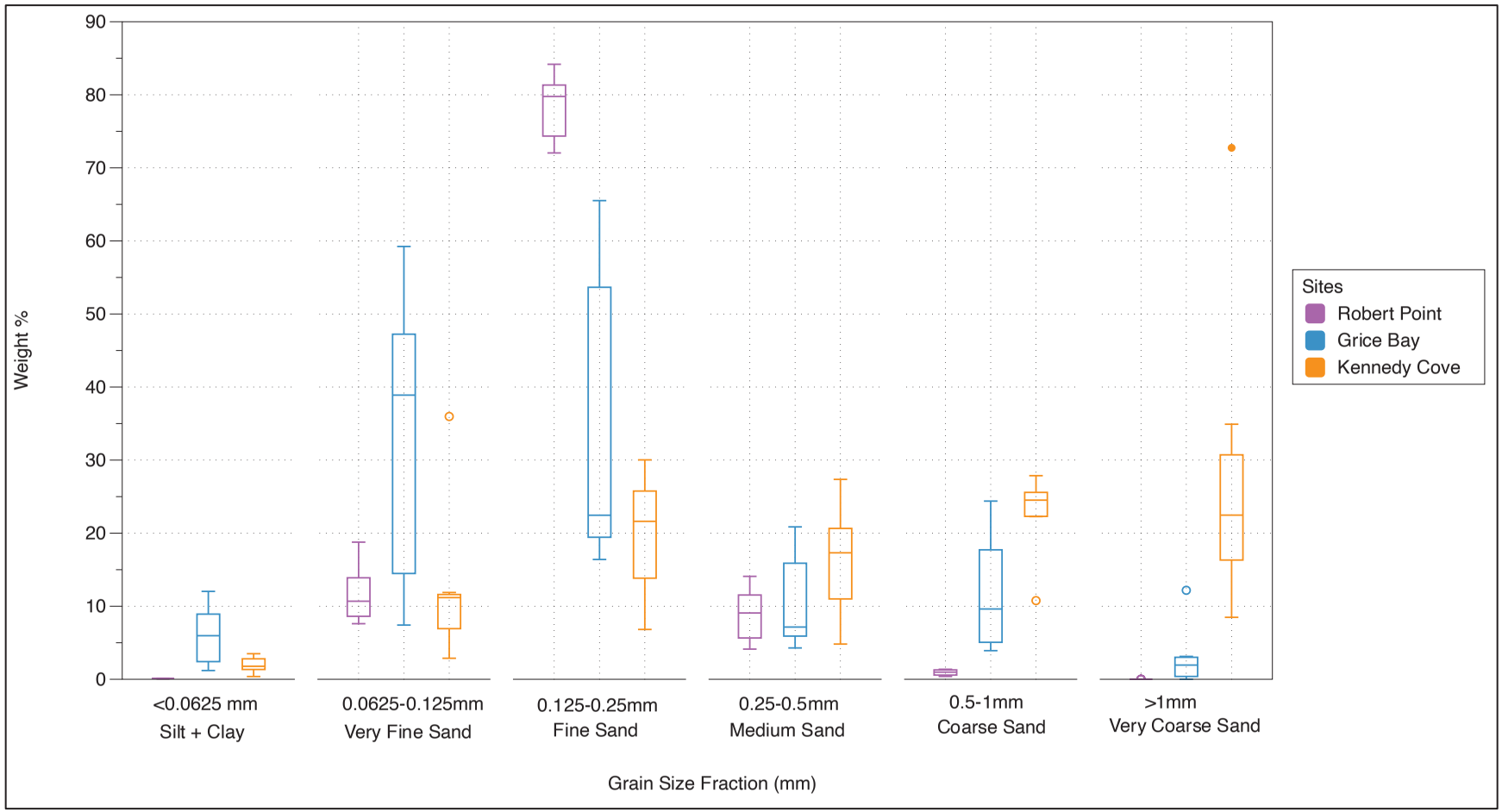


Figure 2. Box and whisker plots showing sediment grain size distribution at Robert Point (RP REF), Grice Bay (GB REF) and Kennedy Cove (KC 3). Differences in grain size distribution for %fine sand and %coarse sand size fractions were found to be statistically significant between the three study sites (%fine: Chi square=15.47, p=0.0004*, df=2; %coarse: Chi square=17.82, p=0.0001*, df=2).

To examine the relationship between sediment grain size distribution and MP concentrations and MPARs between the three sites, we compared the mean (+/- SE) %fine (Figure 3) and the mean (+/- SE) %coarse (Figure 4) sediment fractions against mean (+/- SE) MP concentrations and mean (+/- SE) MPARs for the top 10cm of the cores. We found significant differences in the %fine sediment fraction and MP concentrations between the Kennedy Cove and Robert Point sites (Z=5.188, p=0.040), with Robert Point containing both the highest fraction of fine sediments and highest concentrations of MPs. When comparing the %fine sediment fraction and MPARs, significant differences were observed between all three locations: Grice Bay and Kennedy Cove (Z=9.430, p=0.0089), Robert Point and Kennedy Cove (Z= 13.858, p=<0.0001), and Robert Point and Grice Bay (Z=22.705, p=0.0003). No significant differences were observed with the %coarse sediment size fraction and MP concentrations or MPARs between sites.

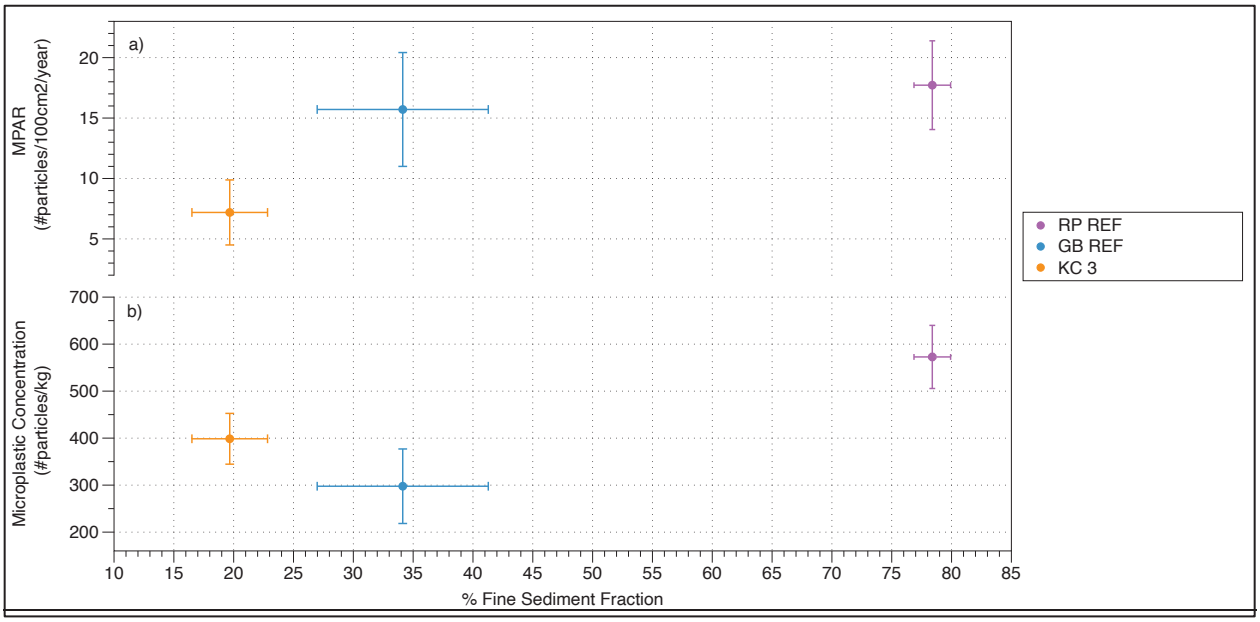


Figure 3. The %fine sediment fraction plotted against a) mean (+/- SE) MP concentration (#particles/kg) and b) mean (+/- SE) MPARs (#particles/100cm^2/year) for Robert Point (RP REF), Grice Bay (GB REF), and Kennedy Cove (KC 3) for the top 10cm of the core. Fine sediment grain size fraction includes fine sand (0.125-0.25mm). Error bars indicate standard error. Significant differences in paired data observed between Robert Point and Kennedy Cove for MP concentrations. Significant differences observed in paired data between all sites for MPARs. 3a) KC3:RP REF (Z= 5.188, p=0.040), KC3:GB REF (Z=3.165, p=0.099), GB REF:RP REF (Z=0.1836, p=0.6748). 3b) KC3:RP REF (Z= 13.858, p=<0.0001), KC3:GB REF (Z=9.430, p=0.0089), GB REF:RP REF (Z=22.705, p=0.0003).

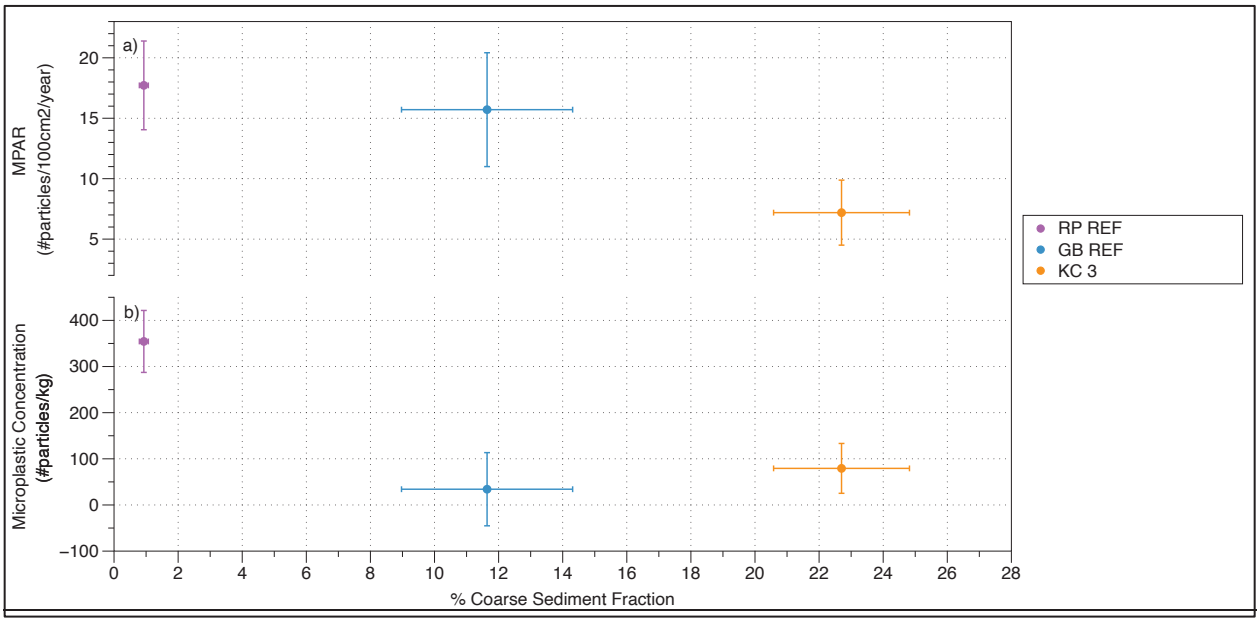


Figure *4.* The %coarse sediment fraction plotted against a) mean (+/- SE) MP concentration and b) mean (+/- SE) MPAR for Robert Point (RP REF), Grice Bay (GB REF), and Kennedy Cove (KC 3) for the top 10cm of the core. The %coarse sediment grain size fraction includes coarse sand 0.5-1mm. Error bars indicate standard error. No significant differences observed in paired data between sites. 4a) KC3:RP REF (Z= 2.388, p=0.146), KC3:GB REF (Z=2.398, p=0.145), GB REF:RP REF (Z=0.0006 p=0.980). 4b) KC3:RP REF (Z= 2.501, p=0.138), KC3:GB REF (Z=0.129, p=0.725), GB REF:RP REF (Z=2.215 p=0.159).

The $\%C_{org}$ was lowest at Robert Point, followed by Grice Bay, and highest at Kennedy Cove, the eelgrass meadow site (Figure 5). While $\%C_{org}$ was significantly different between the three sites (Chi square =61.35, p=0.0001, df=2), we note that $\%C_{org}$ ranged only from 0-2.29% across all sites, and there were no significant differences observed in the relationship between $\%C_{org}$ and MP concentrations or MPARs between any of the study sites (Figure 6).

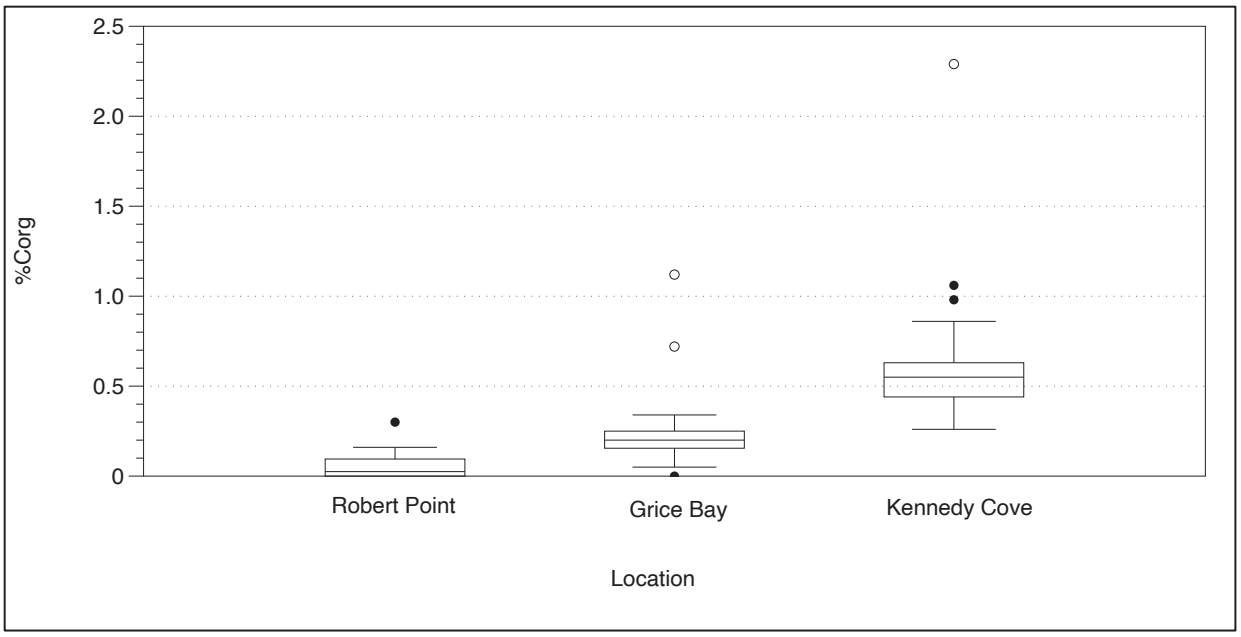


Figure 5. Box and whisker plots showing $\%C_{org}$ at Robert Point (RP REF), Grice Bay (GB REF), and Kennedy Cove (KC 3). $\%C_{org}$ was significantly different between the three sites (Chi square =61.35, p=0.0001, df=2).

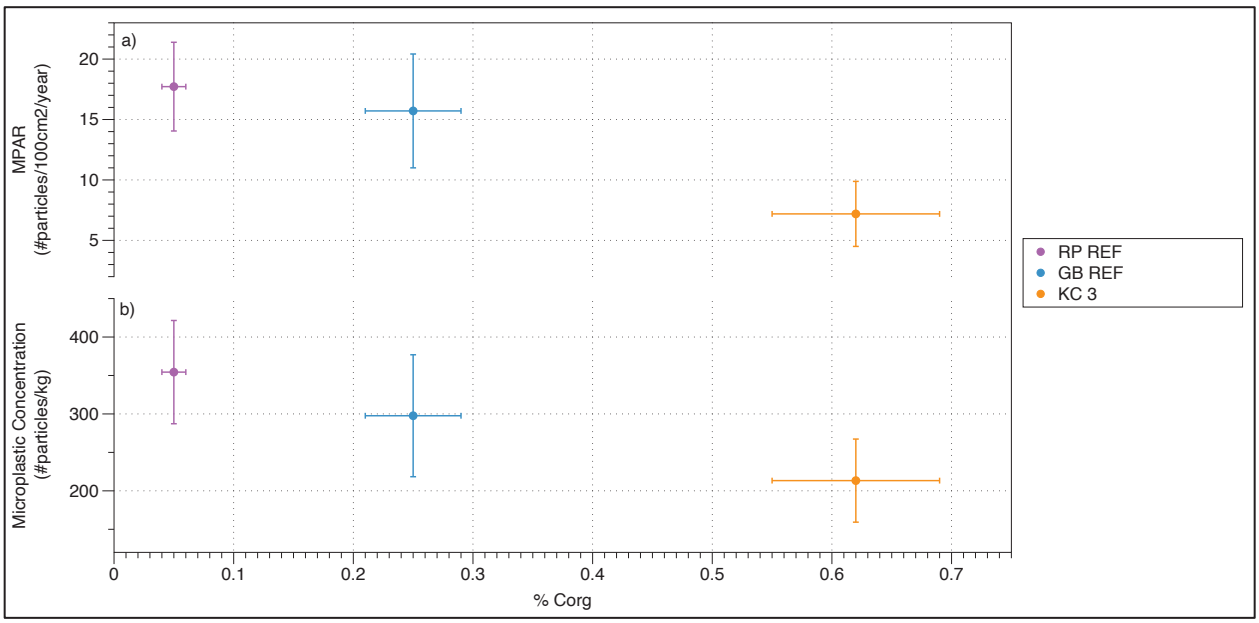


Figure 6. $\%C_{org}$ plotted against a) mean (+/- SE) MP concentration and b) mean (+/- SE) MPARs for Robert Point (RP REF), Grice Bay (GB REF), and Kennedy Cove (KC 3) for the top 10cm of core. Error bars indicate standard error. No significant differences observed in paired data between sites. 6a) KC3:RP REF (Z= 2.671, p=0.112), KC3:GB REF (Z=0.765, p=0.393), GB REF:RP REF (Z=0.300, p=0.590). 6b) KC3:RP REF (Z= 3.391, p=0.276), KC3:GB REF (Z=2.204, p=0.145), GB REF:RP REF (Z=0.099, p=0.756).

2.4.2. Microplastic Characteristics

Microfibers were the sole type of plastic particle found in all samples. The MPs displayed a range of colours, including black (61%), blue (23%), grey (6%), green (6%), and red (4%). The distribution of MP colours is similar across sites (Figure 7). The MPs appeared as thin, often weathered, fibrous strands ranging in length from 0.23-4.92 mm, with a mean (+/- SE) size of 1.77 $\pm$ 0.35 mm and a median size of 1.49 mm (Figure 8). There were no statistically significant differences in MP sizes between sites.

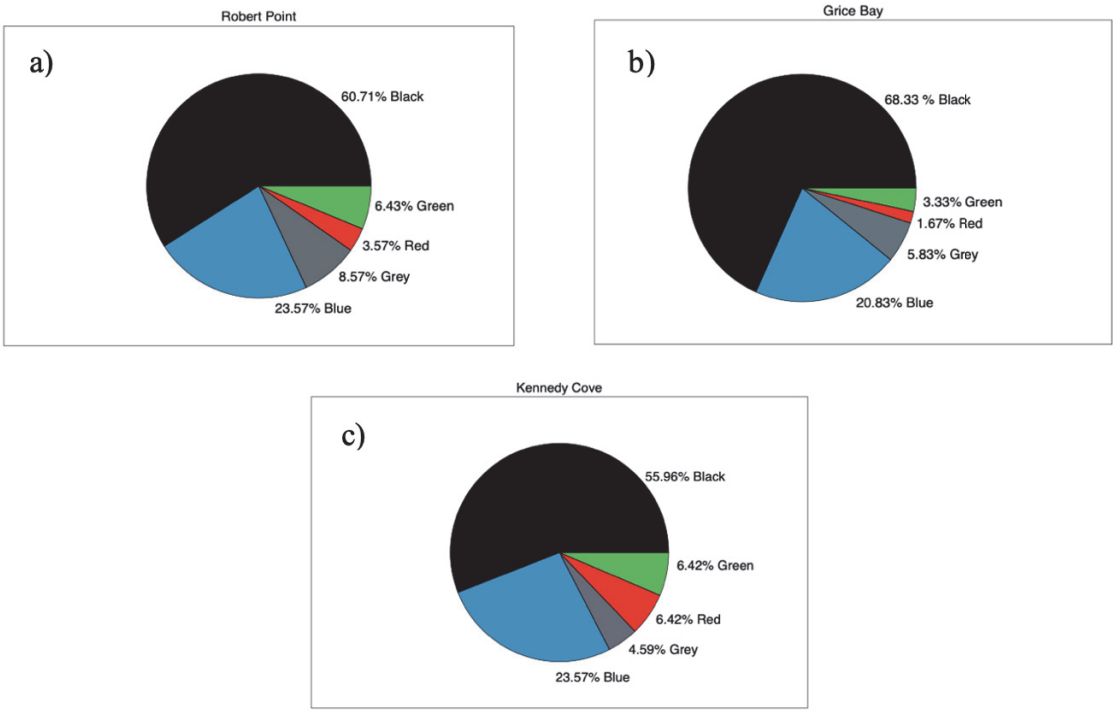


Figure 7. MP colour distribution at a) Robert Point (RP REF), b) Grice Bay (GB REF) and c) Kennedy Cove (KC3).

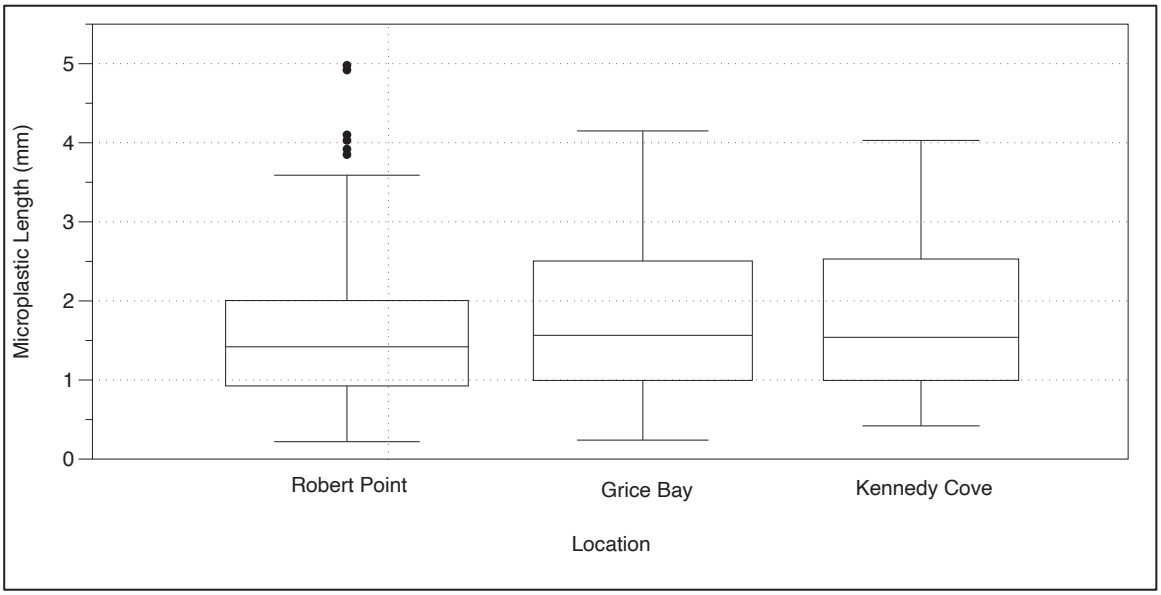


Figure 8. Box and whisker plots showing microplastic length (mm) at Robert Point (RP REF), Grice Bay (GB REF), and Kennedy Cove (KC 3). MP length was not significantly different between sites (Chi square = 1.280, p=0.572, df=2).

2.4.3. Microplastic Concentrations

Across all sites, MP concentrations ranged from 0 - 877 particles/kg (Figure 9). MP concentrations were not significantly different between sites (Chi square = 1.921, p= 0.383, df=2).

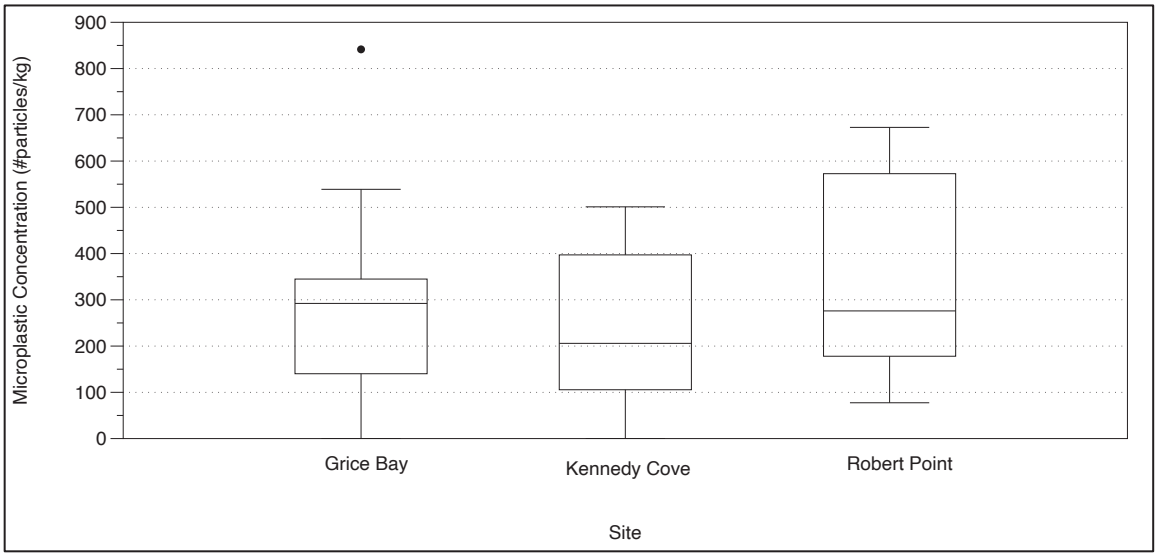


Figure 9. Box and whisker plots showing microplastic concentration (#particles/kg) at Robert Point (RP REF), Grice Bay (GB REF) and Kennedy Cove (KC 3). MP concentrations were not significantly different between sites (Chi square = 1.921, p= 0.383, df=2).

MPs were not evenly distributed throughout the core, and a high degree of variability was observed at depth (Figure 10). While MPs were detected at all depths prior to correction for contamination, after correcting for contamination, MP concentrations at some depths fell to 0 particles/kg. While correction for contamination explains the large number of zeroes, MP concentrations are still variable at depth. This variability could be attributed to oceanographic conditions such as high energy events or strong tidal action, or biological mechanisms of vertical sediment distribution such as bioturbation that rework MPs in the sediment column (Harris, 2020; Loughlin et al., 2021; Yu et al., 2020).

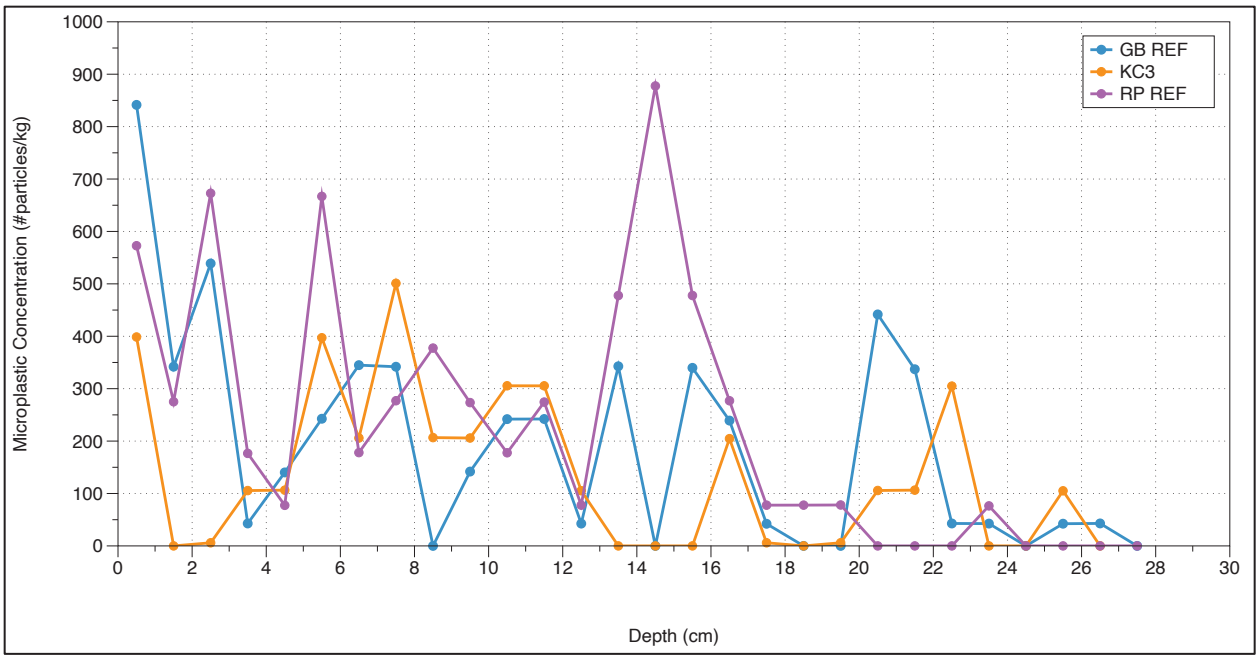


Figure 10. Microplastic concentration (#particles/kg) at depth at Grice Bay (GB REF), Kennedy Cove (KC 3), and Robert Point (RP REF) in Clayoquot Sound. All data corrected for contamination.

MP concentrations hover around 0 particles/kg until about 1950, and then are seen to steadily increase until reaching maximum concentrations in the last 30 years (Figure 11a-c). Based on the timeline during which MPs are being deposited in the sediment, we calculated mean (+/- SE) and median microplastic concentrations for a depth horizon of 10cm (~1980). The highest concentrations of MPs were found at Robert Point with mean and median MP concentrations of 355 ± 119 particles/kg and 276 particles/kg, respectively. The second highest MP concentrations were found at Grice Bay with mean and median MP concentrations of 298 ± 141 particles/kg and 292 particles/kg, respectively. Mean and median MP concentrations at Kennedy Cove were lowest at 213 ± 96 particles/kg and 205 particles/kg, respectively. The mean MP concentration for the entire region was 288 ± 92 particles/kg, with a median concentration of 258 particles/kg. MP concentrations were not found to be statistically different between sites.

2.4.4. Sediment Mass Accumulation Rates (SMARs)

SMARs show an increasing trend starting in the early 20th century, reaching maximum accumulation rates over the last 30 years (Figure 11d-f). SMARs were found to be significantly different between the three sites (Chi square=6.65, p=0.035 df =2).

The highest SMARs were observed at Robert Point (0.01-0.74 g/cm^2/year), followed by Grice Bay (0.13-0.61 g/cm^2/year), and Kennedy Cove (0.01-0.49 g/cm^2/year). We calculated mean (+/- SE) SMARs for the top 10cm of the cores. Robert Point had a mean and median SMAR of 0.51 $\pm$ 0.09 g/cm^2/year and 0.49 g/cm^2/year, respectively. Grice Bay had a mean and median SMAR of 0.50 $\pm$ 0.03 g/cm^2/year and 0.50 g/cm^2/year, respectively. Kennedy Cove had a mean and median SMAR of 0.30 $\pm$ 0.08 g/cm^2/year and 0.23 g/cm^2/year, respectively. The mean and median SMAR across all sites was 0.43 $\pm$ 0.07 g/cm^2/year and 0.48 g/cm^2/year.

2.4.5. Microplastic Accumulation Rates (MPARs)

MPARs across all sites ranged from 0 – 51 particles/100cm^2/year and showed substantial changes over time (Figure 11g-i). MPARs are zero at the oldest depths in all three cores, increase in the 1950s, and reach the highest MPARs in the last 30 years. MPARs were significantly different between the three sites (Chi square = 7.49, p=0.02, df=2). We calculated mean (+/- SE) and median MPARs over the top 10 cm of the cores (~1980), which is comparable to the depths sampled in other studies and the depths over which most MPs are present (Figure 10g-i). Mean MPARs in the region increased from 7 $\pm$ 3 particles/100cm^2/year in 1950 to 33 $\pm$ 12 particles/100cm^2/year in 2016. Grice Bay had the highest MPARs ranging from 0-51 particles/100cm^2/year with a mean and median of 16 $\pm$ 8 particles/100cm^2/year and 14 particles/100cm^2/year, respectively. At Robert Point, MPARs ranged from 0- 33

particles/100cm^2/year with a mean and median of 18 ± 6 particles/100cm/year and 16 particles/100cm^2/year, respectively. Kennedy Cove had the lowest MPARs, ranging from 0 – 25 particles/100cm^2/year$^-$ with a mean and median of 7 ± 5 particles/100cm^2/year and 4 particles/100cm^2/year, respectively. Across the entire region, the mean and median MPARs for the top 10 cm of the cores were 13 ± 5 particles/100cm^2/year and 10 particles/100cm^2/year, respectively.

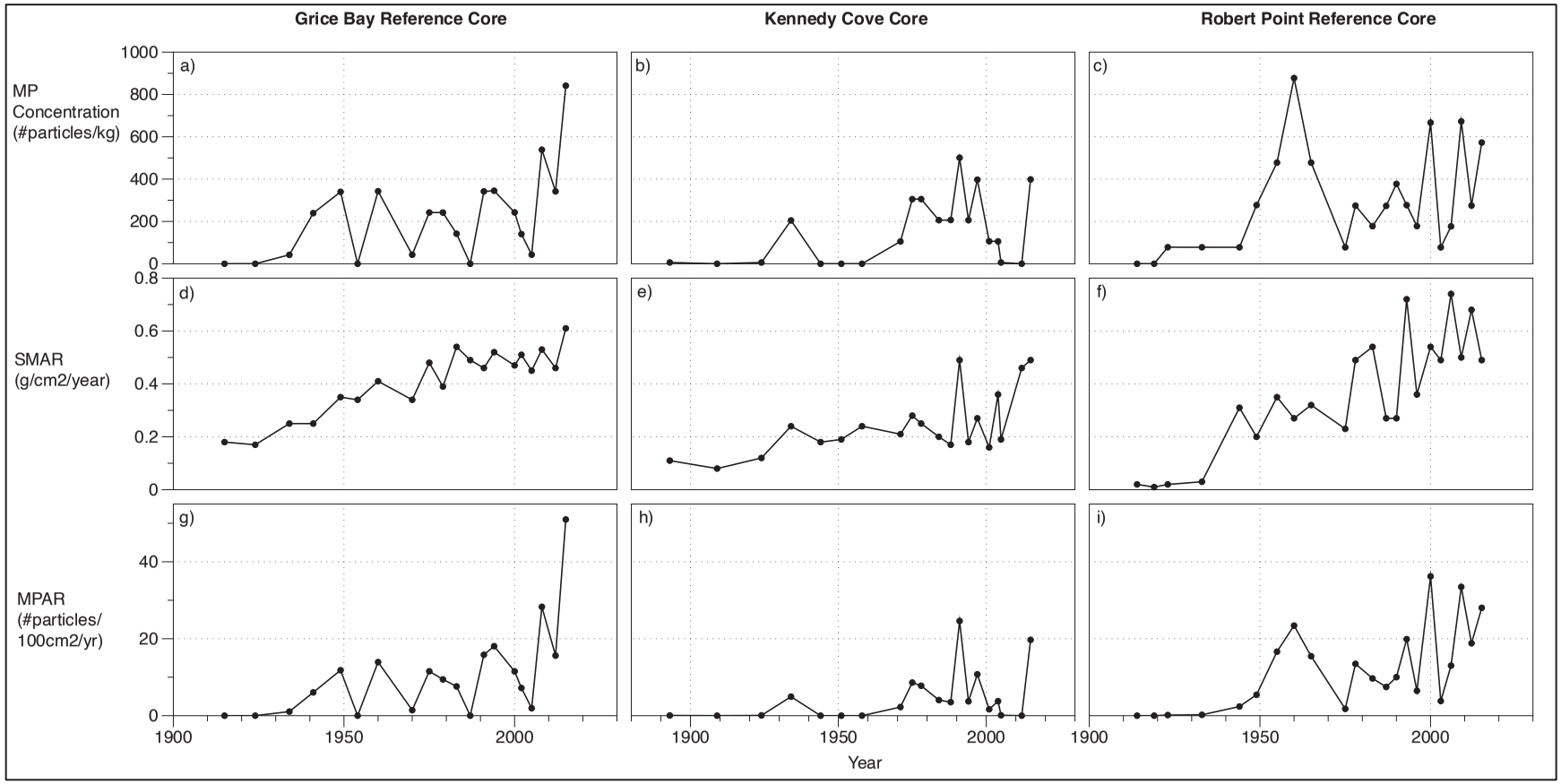


Figure 11. Microplastic concentrations (#particles/kg) (a-c), sediment mass accumulation rates (SMARs) (g/cm^2/year) (d-f), and microplastic accumulation rates (MPARs) (#particles/100cm^2/year) (g-i) at Grice Bay (GB REF), Kennedy Cove (KC 3) and Robert Point (RP REF) vs. Age (Year). All data corrected for contamination.

2.4.6. Global Comparison

A comparison with 70 sites from different marine sedimentary environments across the globe showed that observed mean MP concentrations in Clayoquot Sound were comparable to values from a number of other regions (Figure 12). Of the 70 sites studied, 28 did not take contamination precautions, or report any form of contamination or contamination correction for their study. As contamination has been noted as a major issue with MP studies (Mathalon & Hill, 2014), an additional map was produced removing these sites from the global comparison to highlight the studies that have taken contamination measures (Figure 13). MP concentrations in Clayoquot Sound were averaged over the surface 10 cm of the cores for accurate comparisons with the other studies, putting mean (+/- SE) concentrations in Clayoquot Sound at 288 $\pm$ 92 particles/kg. Similar to Clayoquot Sound, 79% of the 42 sites showed mean and/or median values ranging from 0-500 particles/kg. Across all sites, MPs ranged from 2 – 11,600 particles/kg. Several outliers showed anomalously high MP concentrations including Halifax Harbour, Canada (beach; 5,000 particles/kg), the East Coast of the USA (beach; 1,410 particles/kg), Orkney, Northern Scotland (shallow coastal; 2,300 particles/kg), the Lower Saxonian Wadden Sea (shallow coastal; 11,600 particles/kg), Venice, Italy (lagoon; 1,445 particles/kg), the Tanzanian Coastline (beach; 2,972 particles/kg), Guangdong Coastal Area (shallow coastal; 1,444 particles/kg), Jinjiang Estuary (estuarine; 1,926 particles/kg), the Derwent Estuary (estuarine; 3,315 particles/kg), Baynes Sound (estuarine; 6,903 particles/kg), the Mariana Trench (deep sea; 1,600 particles/kg) and the Hausgarten Observatory in the deep Arctic Ocean (deep sea; 4,356 particles/kg).

Despite differences in environment type, MPs are ubiquitous in marine sedimentary environments and exhibit highly variable concentrations (Figure 14). Beach sediments show the most variable

MP concentrations, while continental shelf sediments consistently have the lowest MP concentrations across all the environment types (Figure 15).

Mean (+/- SE) MPARs at Clayoquot Sound over the last ~30 years were found to be 14 ± 5 particles/100cm^2/year, which were similar in magnitude to mean MPARs observed for the same time period at all other sites studied including the Santa Barbara Basin (28 particles/100cm^2/year), Kuwait Bay (35 particles/100cm^2/year), Tokyo Bay (14 particles/100cm^2/year) and the Rockall Trough (2 particles/100cm^2/year).

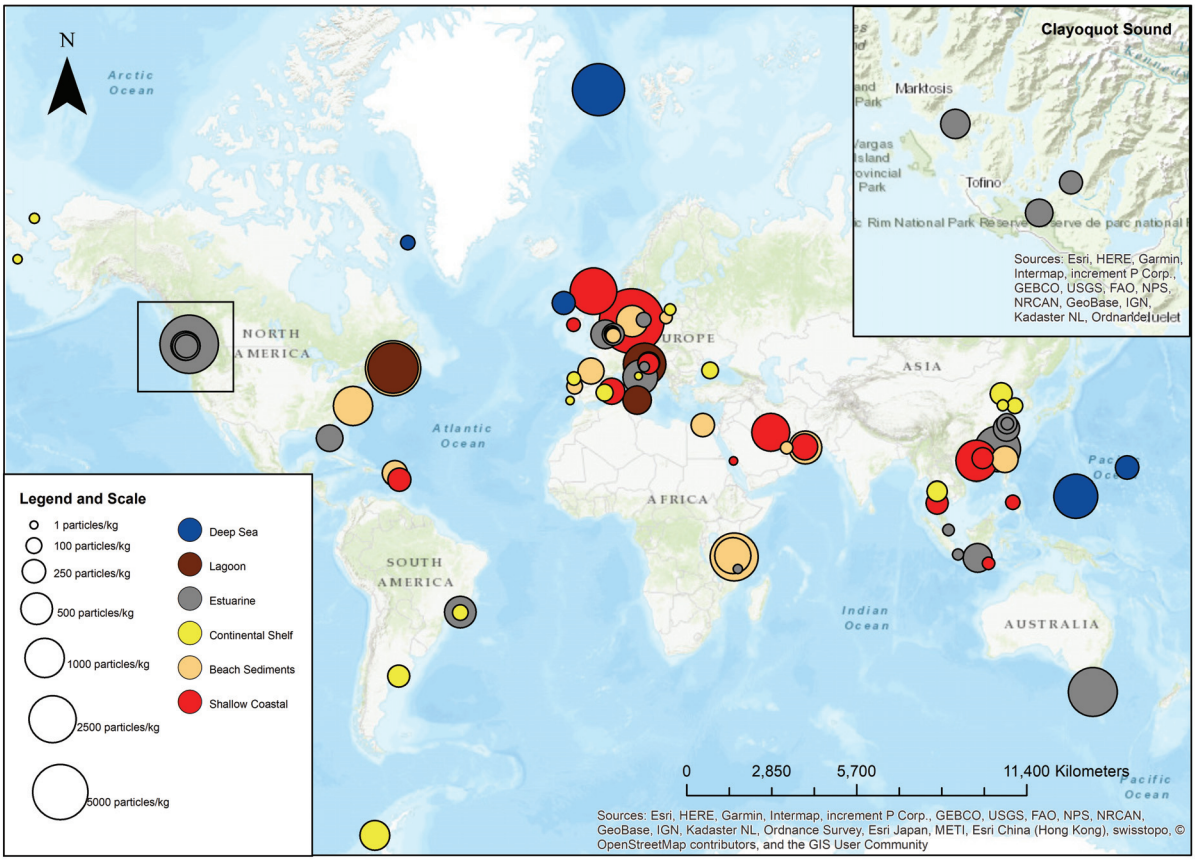
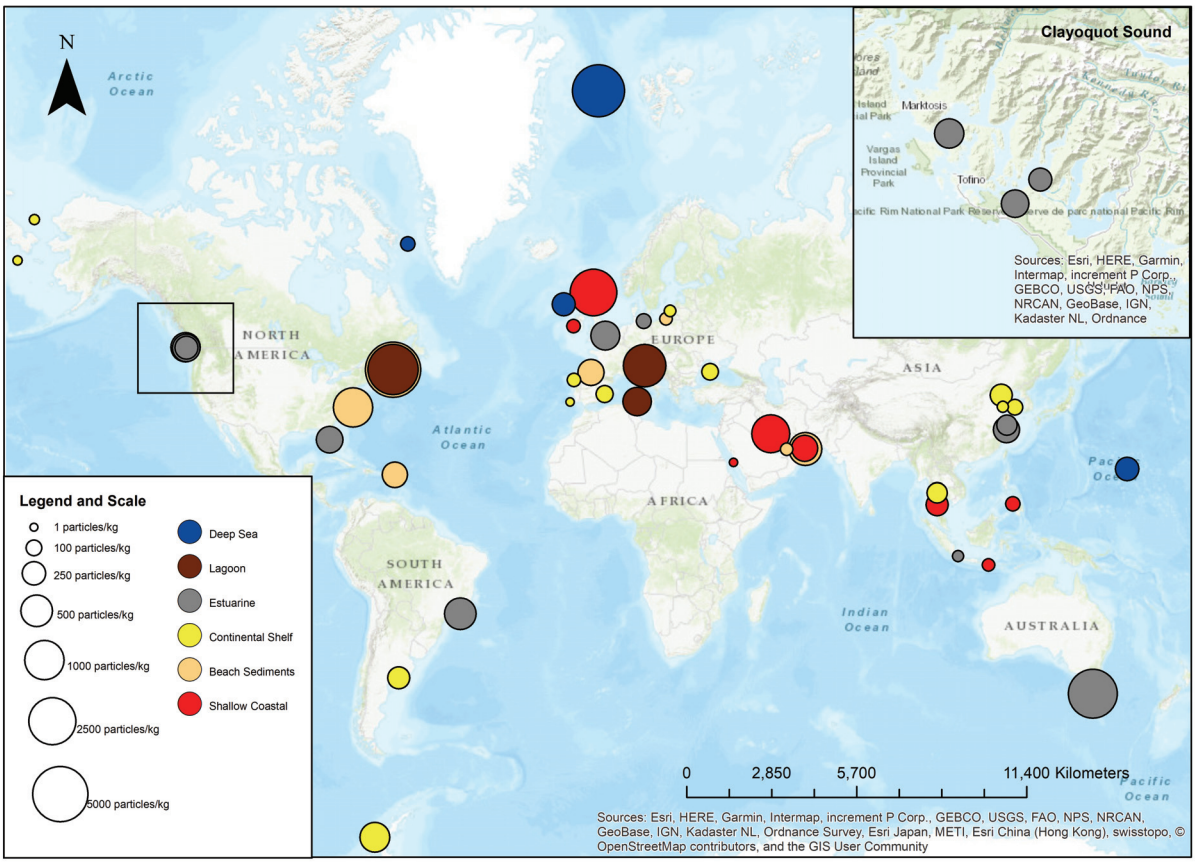


Figure 12. Mean/median microplastic concentrations (#particles/kg) in marine sedimentary environments (beach, shallow coastal, continental shelf, estuarine, lagoon and deep sea) from 70 sites across the globe (not corrected for contamination). (Liebezeit & Dubaish, 2012; Claessens et al., 2011; Mathalon & Hill, 2014; Graca et al., 2017; Bosker

et al., 2018; Dodson et al., 2020; Aslam et al., 2020; Kor et al., 2020; Masiá et al., 2019; Chen & Chen, 2020; Shabaka et al., 2019; Chouchene et al., 2021; Mayoma et al., 2020; Laglbauer et al., 2014; Alomar et al., 2016; Blumenröder et al., 2017; Cordova et al., 2018; Yizheng Li et al., 2021; Pagter et al., 2020; Kor et al., 2020; Al-Lihaibi et al., 2019; Tsang et al., 2017; Akhbarizadeh et al., 2017; Matsuguma et al., 2017; Sandre et al., 2019; Bucol et al., 2020; Mohamed Nor & Obbard, 2014; Peng et al., 2017; Alves & Figueiredo, 2019; McEachern et al., 2019; Firdaus et al., 2020; Yubo Li et al., 2020; Deng et al., 2020; Cordova et al., 2021; Fraser et al., 2020; Horton et al., 2017; Willis et al., 2017; Stolte et al., 2015; Guerranti et al., 2017; Atwood et al., 2019; Kazmiruk et al., 2018; Vianello et al., 2013; Abidli et al., 2018; Zhao et al., 2018; Zheng et al., 2020; Filgueiras et al., 2019; Ronda et al., 2019; Wang et al., 2020; Mistri et al., 2020; Mu et al., 2019; Cincinelli et al., 2021; Carretero et al., 2021; Frias et al., 2016; Zobkov & Esiukova, 2017; Reed et al., 2018; Baptista Neto et al., 2019; Peng et al., 2018; Zhang et al., 2020; Kanhai et al., 2019; Courtene-Jones et al., 2020; Bergmann et al., 2017).



Sources: Abidli et al., 2018, Alomar et al., 2016, Alves & Figueiredo, 2019, Aslam et al., 2020, Bergmann et al., 2017, Blumenröder et al., 2017, Bosker et al., 2018, Chen & Chen, 2020, Claessens et al., 2011, Cordova et al., 2018, Courtene-Jones et al., 2020, Filgueiras et al., 2019, Firdaus et al., 2020, Graca et al., 2017, Haave et al., 2019, Kor et al., 2020, Liebezeit & Dubaish, 2012, Masiá et al., 2019, Mathalon & Hill, 2014, McEachern et al., 2019, Mistri et al., 2020, Mohamed Nor & Obbard, 2014, Peng et al., 2017, Ronda et al., 2019, Vianello et al., 2013, Wang et al., 2020, Zhao et al., 2018, Zheng et al., 2020.

Projection: WGS 1984

Production: Hasini Basnayake, 2021

Figure 13. Mean/median microplastic concentration (#particles/kg) in marine sedimentary environments (beach, shallow coastal, continental shelf, estuarine, lagoon, and deep sea) from 42 sites across the globe (corrected for contamination). (Mathalon & Hill, 2014; Graca et al., 2017; Bosker et al., 2018; Dodson et al., 2020; Aslam et al., 2020; Kor et al., 2020; Masiá et al., 2019; Blumenröder et al., 2017; Cordova et al., 2018; Pagter et al., 2020; Kor et al., 2020; Al-Lihaibi et al., 2019; Akhbarizadeh et al., 2017; Matsuguma et al., 2017; Bucol et al., 2020; Peng et al., 2017; Alves & Figueiredo, 2019; McEachern et al., 2019; Cordova et al., 2021; Fraser et al., 2020; Horton et al., 2017; Willis et al., 2017; Stolte et al., 2015; Vianello et al., 2013; Abidli et al., 2018; Mathalon & Hill, 2014; Zhao et al., 2018; Zheng et al., 2020; Filgueiras et al., 2019; Ronda et al., 2019; Wang et al., 2020; Mu et al., 2019; Cincinelli et al., 2021; Carretero et al., 2021; Frias et al., 2016; Zobkov & Esiukova, 2017; Reed et al., 2018; Zhang et al., 2020; Kanhai et al., 2019; Courtene-Jones et al., 2020; Bergmann et al., 2017).

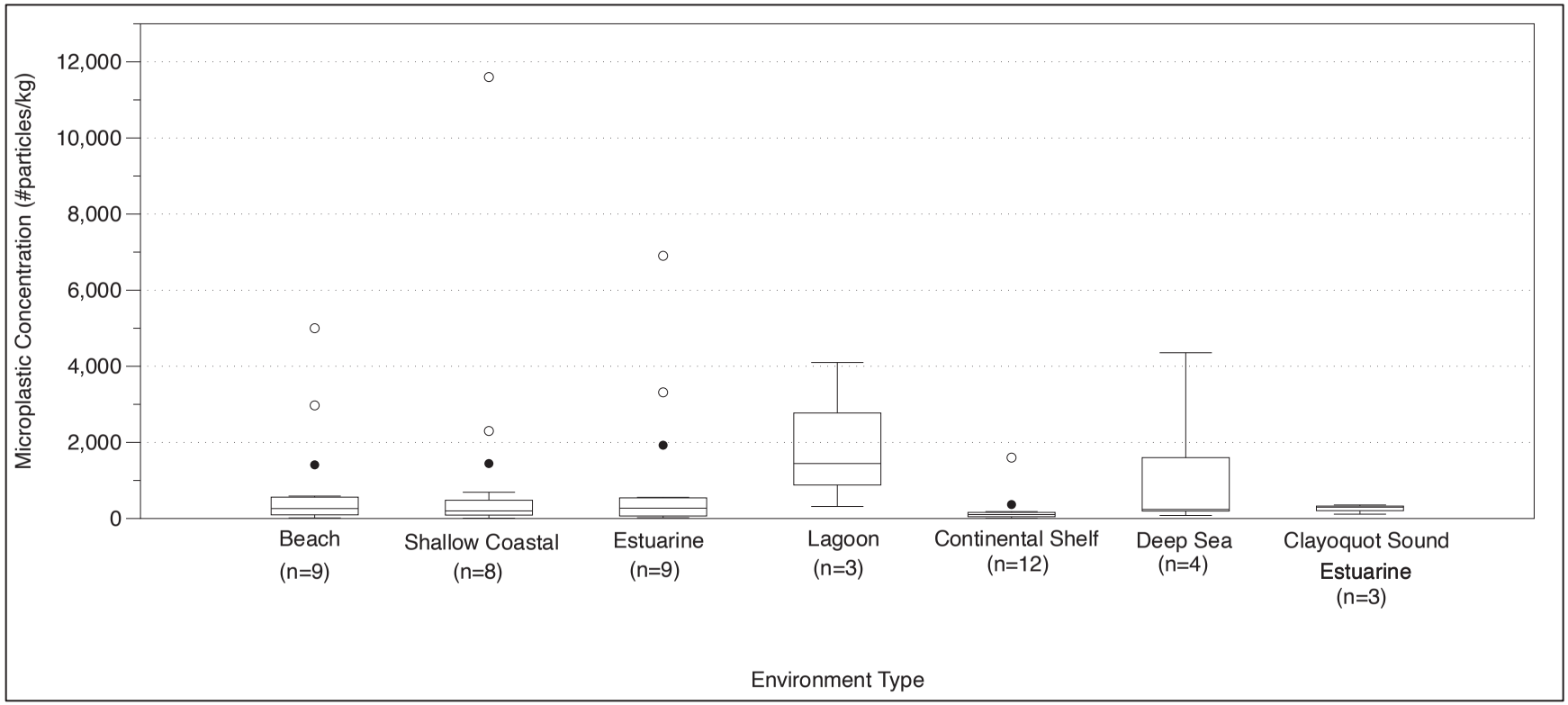


Figure 14. Box and whisker plots showing the distribution of mean microplastic concentrations from marine sedimentary environments across the globe from 42 sites including beach (n=9), shallow coastal (n=8), estuarine (n=9), lagoon (n=3), continental shelf (n=12) and deep-sea (n=4) environments (all corrected for contamination). Mean concentrations for Clayoquot Sound (n=3) are also included.

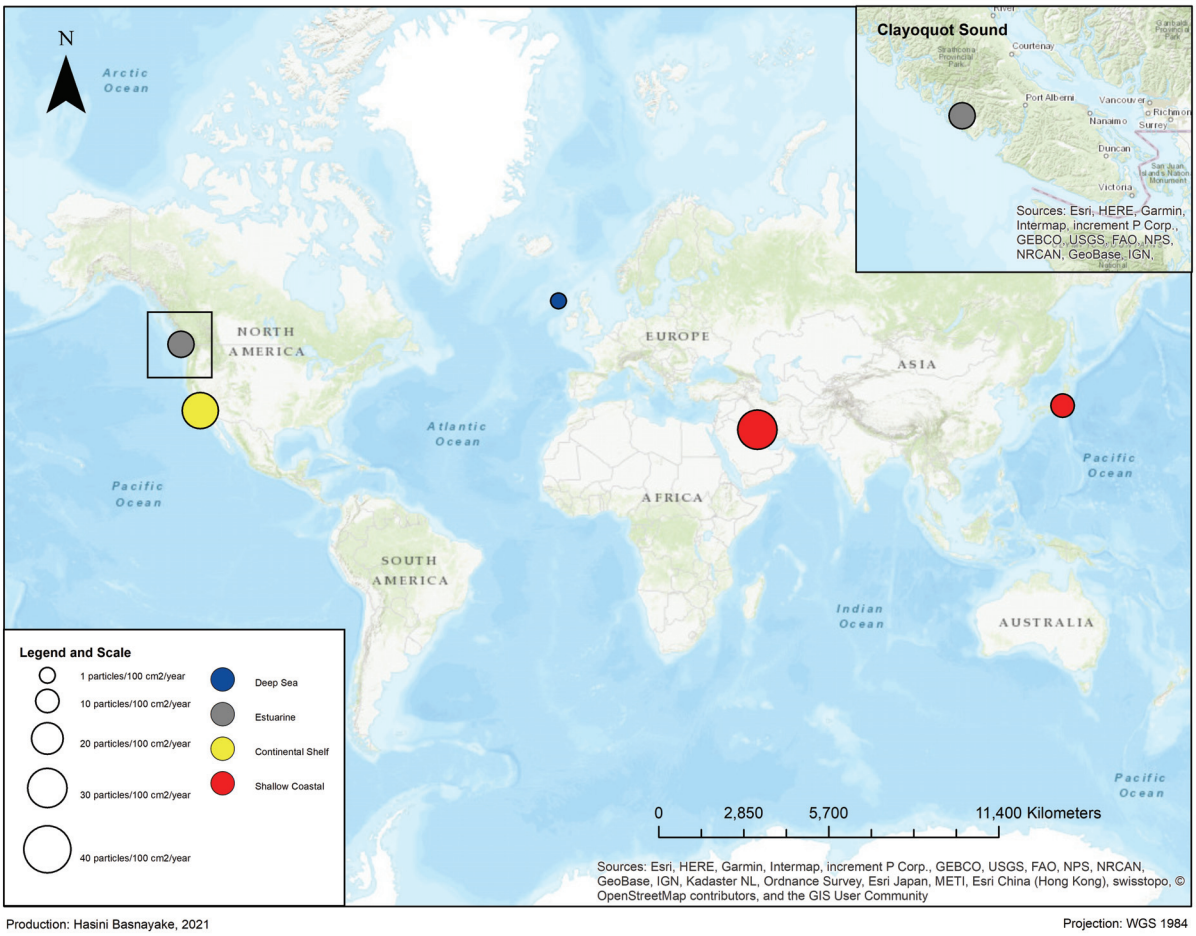


Figure 15. Mean microplastic accumulation rates (particles/100cm^2/year) over the last ~30 years from different marine sedimentary environments (shallow coastal, continental shelf, estuarine, and deep sea) from across the globe (corrected for contamination). (Brandon et al., 2019; Courtene-Jones et al., 2020; Matsuguma et al., 2017; Uddin et al., 2021).

2.5. Discussion

2.5.1. Ubiquity of Microplastics in Clayoquot Sound

MPs were present in all three cores collected from Clayoquot Sound, pointing to the ubiquity of MPs in marine sedimentary environments. Our results corroborate those of Harris (2020) who found that only 5% of 80 studies from across the globe reported samples or replicate samples in

which MPs were not detected. In our study, MPs were detected in 100% of samples prior to correcting for contamination and in 85% of samples after normalizing for contamination. The results of our study show that MPs are not only present in densely populated regions, but also in more remote regions such as Clayoquot Sound on the west coast of Vancouver Island. We note, however, that although Clayoquot Sound is remote in terms of human population and limited accessibility, our sites are near other human sources of MPs, such as aquaculture and wastewater effluent. The results of our study provide confirmation that MPs are ubiquitous in the marine sedimentary environment (Alimba & Faggio, 2019; Avio et al., 2017) both at the surface and at depth, regardless of location.

2.5.2. Relationship Between Microplastics and Depositional Environment

Depositional environment plays an important role in the fate of MPs in marine sedimentary environments (Harris 2020; Enders et al., 2019). Studies on MPs in the water column have shown a density stratification of MPs, with the densest MPs settling into the sediments (Gago et al., 2017; Li et al., 2021; Desforges et al., 2014). In the region of Clayoquot Sound, estuarine fjord-style inlets create a dynamic environment with freshwater flowing into a marine environment resulting in a variety of horizontal and vertical density structures that could influence the stratification of MPs (Okey & Dallimore, 2015). Strong upwelling that occurs in the summer months may also be responsible for delivering MPs trapped in deep sea environments, which are known sinks for MPs, to the study sites (Okey & Dallimore, 2015; Bergmann et al., 2017). At the sediment-water interface, strong tidal currents and high energy events common in the area could result in redistribution and resuspension of MPs into the water column (Okey & Dallimore, 2015; Coote, 1964). The Kennedy Cove site is unique and while the river is unlikely to deposit large amounts of MPs, the outflow of the Kennedy River may have an impact on MP deposition by altering the depositional environment at the site. In the sediment column, highly variable MP concentrations at depth in Clayoquot Sound may point to bioturbation or other processes of vertical mixing that

are known to redistribute sediments (Loughlin et al., 2021). In our study we examined certain sedimentary properties as indicators of depositional environment at the sites, and to diagnose the relationship between MP deposition and sedimentary properties.

To understand the relationship between sedimentary properties and MP deposition, we measured sediment grain size distribution and $\%C_{org}$. We observed significant differences in $\%C_{org}$ between sites, as well as in the %fine sediment size fraction and %coarse sediment size fraction between sites. Robert Point contained the highest percentage of fine sediments (78%), and lowest $\%C_{org}$, indicative of a lower energy environment. Kennedy Cove had the highest percentage of coarse sediments (25%), indicative of a higher energy environment, but also possessed the highest $\%C_{org}$. The coarser sediments at Kennedy Cove could be explained by sediment deposition from the Kennedy River. The elevated $\%C_{org}$ could result from the fact that the core was taken further out in the subtidal zone in the eelgrass meadow. In contrast, the Robert Point and Grice Bay cores were taken in the intertidal zones.

The relationship between $\%C_{org}$ and MP deposition is unclear, and the literature is divided with some studies suggesting a relationship between $\%C_{org}$ and MP deposition (Haave et al., 2019; Maes et al., 2017; Vianello et al., 2013), while others suggest there is no relationship (Alves & Figueiredo, 2019; Courtene-Jones et al., 2020; Mu et al., 2019; Ronda et al., 2019). Naturally occurring organic matter has a density similar to plastic (~ 0.9-1.4 g/cm^3) and thus MPs might behave similarly to organic matter and settle in areas with high $\%C_{org}$ in the marine sedimentary environment (Enders et al., 2019). The studies that noted a relationship between $\%C_{org}$ and MP deposition found that MPs were accumulating in areas with higher $\%C_{org}$ in lower energy environments (Courtene-Jones et al., 2020; Mu et al., 2019; Vianello et al., 2013). Bergmann et al., (2017) found that MP concentrations correlated with Chlorophyll A and particulate organic carbon (POC), and Kvale et al. (2020) showed that organic material, including marine snow and animal fecal pellets, play a role in MP settling in the deep sea. In Clayoquot Sound, $\%C_{org}$ was

found to be statistically different between the sites, but when compared with MP concentrations and MPARs there were no statistically significant differences observed between sites. We note that $\%C_{org}$ was low (<2.9%) across all sites, though previous studies that found significant relationships between organic carbon content and MP concentrations in marine sediments in Italy and the Northeast Atlantic Ocean also exhibited low organic carbon content (< 3.5%). It is possible that incorporation of sites with higher $\%C_{org}$ would provide more clarity on the relationship between $\%C_{org}$ and MP deposition, but our current results suggest no statistically significant link.

Sediment grain size is another property linked with MP deposition, as fine grained sediments have similar densities to MPs and are typically found in low energy environments (Enders et al., 2019). Similarly to $\%C_{org,}$ the literature is divided on whether a relationship exists between sediment grain size distribution and MP deposition, with some studies claiming that a relationship exists (Enders et al., 2019; Haave et al., 2019; Maes et al., 2017; Vianello et al., 2013; Zhang et al., 2019), while others suggest there is no clear relationship (Alomar et al., 2016; Filgueiras et al., 2019; Mohamed Nor & Obbard, 2014; Wang et al., 2020). The studies that noted a relationship found higher concentrations of MPs in the finer sediment size fractions (Maes et al., 2017; Vianello et al., 2013; Zhang et al., 2019). The Kennedy Cove core, with the coarsest sediments and lowest MP concentrations may be influenced by the outflow of the Kennedy River, creating a higher energy environment with larger grain size fractions in which we see less MP deposition. The sites in Clayoquot Sound show a significant, positive relationship between MP concentration and MPARs and the %fine sediment fraction, supporting the literature which states that MPs tend to be deposited in finer sediment associated with lower energy environments.

2.5.3. Sources of Microplastics

Based on the movement of surface currents around Vancouver Island, one likely source of MPs to Clayoquot Sound is water from the industrialized Strait of Georgia and Puget Sound, which

reaches the outer coast of Vancouver Island through the Juan de Fuca Strait and the VICC. Desforges et al. (2014) found that mean MP concentrations in subsurface waters (0-4.5m) of the northeastern Pacific Ocean, including the waters of the Alaskan and California currents, were several orders of magnitude greater than in other regions of the North Pacific Ocean, including the North Pacific Gyre, the west coast of the United States, and the Bering Sea (Desforges et al., 2014). Furthermore, MP concentrations were 4-27 times greater nearshore, in the VICC, compared with offshore sites situated in the Alaskan and California Currents (Desforges et al., 2014). Fibers made up 70% of the identified plastics in coastal waters, and the contribution of fibers to total MPs decreased with increasing distance from shore (Desforges et al., 2014). One likely source of high MP concentrations in these coastal waters is the Salish Sea estuary, into which several highly populated areas (e.g., Vancouver, Victoria and Seattle) release their wastewater effluent. Waters from the Strait of Georgia, Juan de Fuca Strait, and Puget Sound then flow westward through the Juan de Fuca Strait and along the coast within the VICC in summer and the Alaskan Current in winter.

The similarity between the MP fibers in Clayoquot Sound sediments and subsurface water samples off the coast of Vancouver Island (Desforges et al., 2014) suggests that coastal waters in the Strait of Georgia and Puget Sound are an important source of MPs. Clayoquot Sound is made up of fjord-type inlets, with shallow sills and narrow connecting passages which restrict connection to the open ocean to that of the upper 11-13 meters of the oceanic surface layer (Coote, 1964). Thus, it is likely that MPs enter Clayoquot Sound in surface waters from the outer coast, are transported through the Calmus Passage, Browning Passage, and Father Charles Channel, and remain suspended in the surface waters due to tidal activity before being deposited in Clayoquot Sound sediments.

Wastewater effluent is likely another source of MPs. Multiple studies report wastewater treatment plants (WWTP) as a major source of MPs to the marine environment, and the vast majority of

these studies have found microfibers, particularly polyester, to be the dominant MP found in the WWTPs (Dris et al., 2016; Sun et al., 2019; Talvitie et al., 2017; Ziajahromi et al., 2017). In the Tofino Inlet, untreated wastewater is discharged directly into Duffin Passage near the mouth of the Inlet (District of Tofino, 2017) (Figure 1). From there, ebb currents flowing northeasterly into the Inlet during flood tides likely carry MPs from the WWTP in Tofino directly to Grice Bay and Kennedy Cove. In the nearby Salish Sea, a secondary WWTP serving 1.3 million people near the City of Vancouver reported an estimated 1.76 trillion MPs entering the WWTP every year, with 98% being retained and 30 billion MPs being released into the receiving environment (Gies et al., 2018). The study also found 70% of the MPs in the WWTP to be microfibers (Gies et al., 2018). Thus, MPs from WWTP effluent from the Salish Sea could also reach Clayoquot Sound via the oceanographic currents described above.

A final important source of MPs could be local aquacultural development, which has increased from two aquaculture tenures in 1985 to 17 active tenures today accounting for 855ha of ocean in Clayoquot Sound (Province of British Columbia, 2021). For reference, Clayoquot Sound encompasses an area of 350,000 hectares, of which 87,000 hectares is ocean and lake (Province of British Columbia, 2021). Many MP studies have suggested that rope, fishing gear, nets, and waste from aquaculture activities contribute a substantial portion of MPs to seawater and underlying sediments (Covernton et al., 2019; Krüger et al., 2020; Lusher et al., 2017; Mathalon & Hill, 2014). For example, common polymer ropes in a sublittoral marine environment have been shown to lose between 0.39% and 1.02% of their mass per month (Welden & Cowie, 2017). Furthermore, studies have found higher concentrations of MPs in sediments closer to fish farms compared to reference zones (Krüger et al., 2020). The MPs described by Krüger et al., (2020) were primarily black fibers that ranged in size from 0.13 -13.40 mm and averaged 1.5mm (Krüger et al., 2020). We also found predominantly black fibers of a similar size (0.23-4.92mm with a mean +/- SE of 1.77 $\pm$ 1.12 mm), suggesting that aquaculture may be responsible for some of the MP

deposition at the sites. Finally, the timing of changes in MPARs in all three locations also suggest that aquacultural expansion could be contributing to MP deposition in Clayoquot Sound. MPARs begin to increase in the 1980s, mirroring increases in aquaculture activities. While difficult to assess accurately based on available DFO data, the timing of MPAR increase aligns with aquaculture development in the region and therefore supports the idea that aquacultural development could contribute to an increase in MPs in the sediments of the region.

2.5.4. Global Comparison

Our global comparison revealed that MPs are ubiquitous, regardless of marine sedimentary environment, as MPs were found in every environment type. Beach sites had notably higher MP concentrations, while continental shelf sites consistently revealed the lowest concentrations of MPs. Beach sediments may show increased MP concentrations as beaches can act as both sources and sinks of MPs (Schröder et al., 2021). It is noteworthy that the continental shelf sites, where Lebreton et al., (2019) described most of the plastics should be settling, had the lowest MP concentrations. Lebreton et al., (2019)'s model is based on the assumption of positively buoyant plastics, not solely MPs, and does not account for variability in the morphology and sedimentary environment of continental shelf sediments, which might explain the discrepancy between their model and the findings of our global comparison.

MPs were also found to be ubiquitous, regardless of whether the study location was urban or remote, with remote sites characterized by low population density, limited accessibility, and little anthropogenic input. Most of the studies researched were done on sediments near densely populated coastlines, but a few sites, namely the Hausgarten Observatory, the Rockall Trough, the Mariana Trench and the Bering and Chuckchi Seas are more remote. These sites possessed some of the highest MP concentrations, with the Mariana Trench and Hausgarten Observatory being among the highest in the comparison. The high concentrations in deep sea sediments may

be attributed to the deep sea acting as a major sink for MPs which has been observed in the literature (Woodall et al., 2014).

Based on the global comparison, we noted certain studies with anomalously high outliers of MP concentrations which may be explained by differences in methodology, namely in the quantification and correction of contamination in the samples. Of the original 70 sites compared, 28 did not take any contamination measures. Of the 12 studies that reported anomalously high MP concentrations, only 5 corrected for contamination. Contamination has been noted as a prominent issue with MP studies (Mathalon & Hill, 2014; Willis et al., 2017; Woodall et al., 2014), capable of inflating MP estimates. The study by Mathalon & Hill (2014) which showed one of the highest concentrations of MPs in our compilation found that MPs circulating throughout the lab were contaminating their samples, and cautioned that absolute MP concentrations from their study should be used with caution. As many of the other studies with anomalously high MP concentrations did not report contamination, it is possible that this explains their higher MP concentrations.

In addition to quantifying MP concentrations in Clayoquot Sound, our study also showed changes in MPARs through time. Only 7 other published studies on MPs in marine sedimentary environments provided chronological control to their sediments, making our study one of the first to look at temporal changes in MP accumulation at depth in marine sedimentary environments. While these sites represent different sedimentary environments, all of the sites show a similar trend of the highest observed MPARs in the surface layers with a decreasing trend at depth. Furthermore, mean MPARs at Clayoquot Sound documented for the last ~30 years are of a similar order of magnitude to MPARs documented at other locations around the globe (Figure 15). In the Santa Barbara Basin, a basin closely linked to urban environments, Brandon et al., (2019) reported MPAR values similar to those found in Clayoquot Sound. Our results corroborate their results and show that even in a relatively remote region, we see similar increases in MPARs

(Figure 16). Understanding temporal changes in MPARs allowed us to make inferences about certain events that coincided with increases in MP accumulation in the sediments. In the sediments of Clayoquot Sound, MPARs are seen to increase drastically in the 1980s, around the same time that the region saw its first aquaculture tenures. Since then, aquaculture in the region has increased 10-fold, which is also reflected in the increases in MPARs in the sediments. Similarly to the Santa Barbara Basin, temporal changes in MPARs in the sediments of Clayoquot Sound are strongly related to increases in global plastic production (Brandon et al., 2019).

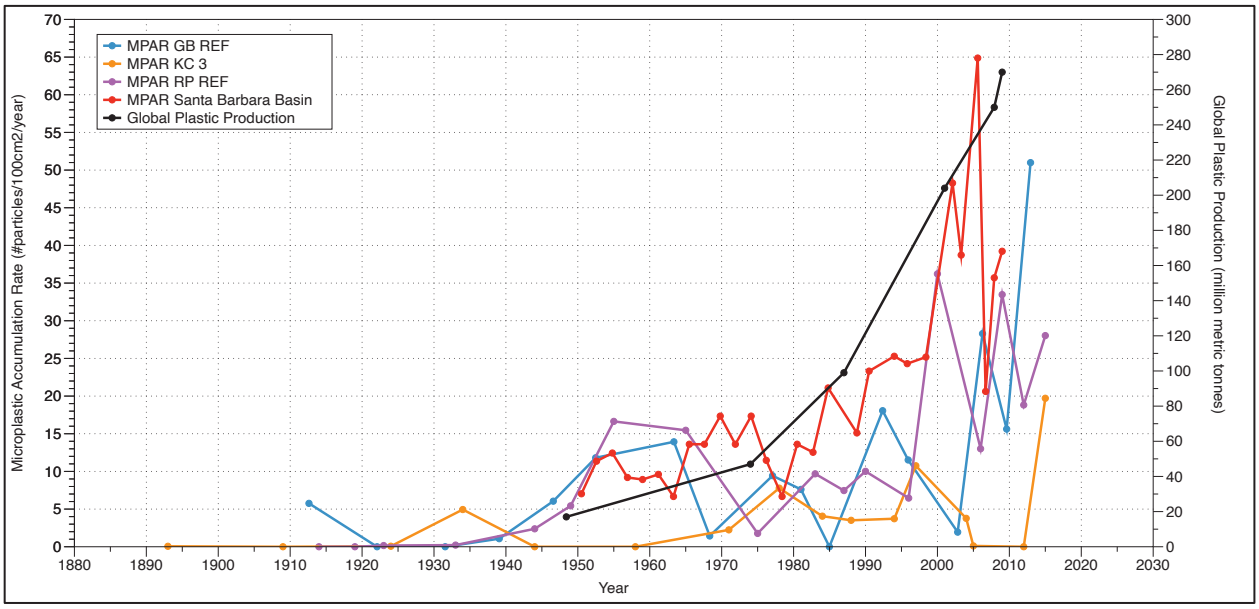


Figure 16. Microplastic accumulation rates (#particles/100cm^2/year) in Clayoquot Sound (Grice Bay (GB REF), Kennedy Cove (KC 3) and Robert Point (RP REF) and the Santa Barbara Basin compared against global plastic production (million metric tonnes).

Interestingly, the accumulation of MPs in the sediments of Clayoquot Sound indicate a plastic horizon in the mid-20th century that reflects the onset of the Anthropocene, an epoch characterized by mass extinctions, polluted oceans, and altered atmosphere among other lasting impacts (Irabien et al., 2020; Lewis & Maslin, 2015). Exponential growth of plastic production has mirrored the post-war socioeconomic growth, industrialization, and environmental degradation associated with the Anthropocene. The Anthropocene has already been observed in the sedimentary record

in other places. In northern Spain, Irabien et al., (2020) found that Holocene and Anthropocene foraminiferal and geochemical contents are distinctly different, with the Anthropocene marked by enhanced concentrations of trace metals and the occurrence of artificial radioisotopes and glass microspheres. Thus, MPs may serve as a geologic proxy in the sedimentary record to denote the onset of the Anthropocene (Brandon et al., 2019). MPARs in the sediments of Clayoquot Sound support this notion, as MP deposition is seen to increase markedly around 1950. MPARs in the post-1950 era are 100 times greater than in the period before 1950. The most drastic increases in MPARs were observed to begin in the 1980s. These data support the idea of a plastic horizon in the mid-20th century marking a significant change in MP deposition across the globe.

2.6. Conclusion

MPs in the environment are a growing concern as myriad studies from across the globe describe the ubiquity of MPs in environmental matrices. Our study supports these findings, with MPs detected at all depths in all sediment cores that we investigated in Clayoquot Sound, British Columbia, Canada. We discovered a relationship between MP concentration and MPARs and the %fine grain size fraction, which corroborates other research that has found higher concentrations of MPs in finer sediments, associated with lower energy environments (Courtene-Jones et al., 2020; Maes et al., 2017; Vianello et al., 2013). While we cannot comment definitively on the sources of MPs to our study sites, the close proximity of the sites to wastewater effluent and aquaculture activities suggests that these are likely sources of MPs to the region. The MPs identified in our study were all microfibers, which corroborates the findings of Desforges et al., (2014) who found predominantly microfibers in the upper surface layers of the Northeastern Pacific Ocean, in the coastal waters adjacent to our study sites. Tidal currents are likely responsible for introducing these marine sources of MPs to the sites. Comparing MP concentrations from our study to studies from across the globe showed that mean (+/- SE) MP

concentrations in the top 10 cm of sediment at our sites (288 ± 92 particles/kg) were on a similar order of magnitude to 79% of sites studied from around the globe, with observed MP concentrations falling between 0-500 particles/kg.

Our study is novel in that it is the first to characterize temporal trends in MP deposition in marine sediments in BC. In his analysis of MPs in different sedimentary environments, Harris (2020) states that in order to assess where MPs are accumulating in the ocean and which habitats are at risk, MPARs are needed, rather than simply MP concentrations. Mean (+/- SE) MPARs in Clayoquot Sound over the last ~30 years were 14 ± 5 particles/100cm^2/year, similar to MPARs documented for marine sedimentary environments across the globe ranging from 2-35 particles/100cm^2/year. Mean (+/- SE) MPARs in Clayoquot Sound increased from 7 ± 3 particles/100cm^2/year in 1950 to 33 ± 12 particles/100cm^2/year in 2016. These increases in MPARs mirror global plastic production, as well as aquaculture development in Clayoquot Sound. The timing of deposition of MPs in the sediments of the region indicate that MPs could serve as a geologic proxy to denote the onset of the Anthropocene, which is believed to begin around the same time that MPs are seen to be deposited in the sediments, around the 1950s (Irabien et al., 2020; Lewis & Maslin, 2015).

Chapter 3. Microplastic deposition in aquatic sediments from urban and remote locations in southwestern British Columbia

3.1. Abstract

Characterizing the distribution of microplastics in different sedimentary environments, from remote and urban locations, is necessary to provide a better understanding of their global distribution, and to comment on their ubiquity, persistence, and potential impacts to biota. In this study we analyzed microplastic concentrations in sediments from two locations from the Metro Vancouver area of southwestern British Columbia (BC), Canada: (1) tidal mud flats in the relatively well-populated area of Boundary Bay, and (2) Orchid Lake, a remote sub-alpine lake in the Seymour Watershed. We selected the study sites to test whether differences in sedimentary environment, as well as location (urban vs. remote), impact microplastic concentrations. We measured sediment grain size distribution and percent organic carbon ($\%C_{org}$) to characterize the sedimentary environments at the sites. We observed microplastics deposited in the top 10 cm of cores at both Boundary Bay and Orchid Lake, and we used this depth horizon to calculate mean microplastic concentration values in the region. Surprisingly, mean microplastic concentrations were 2x greater at Orchid Lake than Boundary Bay. At Boundary Bay, microplastic concentrations ranged from 0-349 particles/kg with a mean (+/- SE) and median concentration of 114 $\pm$ 61 particles/kg, and 50 particles/kg, respectively. At Orchid Lake, microplastic concentrations ranged from 0-908 particles/kg with a mean (+/- SE) concentration of 223 $\pm$ 188 particles/kg. Microplastics were not evenly distributed throughout the core and displayed a high degree of variability at depth, which is likely the result of data correction and sedimentary processes of vertical sediment redistribution. While differences in microplastic concentrations between the sites were not statistically significant, the higher concentrations at Orchid Lake suggest that oceanographic and sedimentary processes at Boundary Bay may be reworking and displacing microplastics in the

sediments. There were no significant relationships detected between sediment properties, including sediment grain size distribution and $\%C_{org}$, and microplastic concentrations. The main sources of microplastics to Boundary Bay are believed to be contaminated water from the Strait of Georgia and Puget Sound, with wastewater effluent being a major contributor to microplastics in these waters. At Orchid Lake, aerial deposition is believed to be the primary source of microplastic deposition. Microplastic concentrations at Orchid Lake may serve as a baseline for aerial deposition of microplastics in the region and could have management implications for microplastics in drinking watersheds. While all types of microplastics were considered, the microplastics found in this study were all microfibers, predominantly black and blue in colour, which corroborates research from around the globe that has found microfibers to be the dominant microplastic in aquatic sedimentary environments.

3.2. Introduction

Understanding the distribution of microplastics (MPs) in different aquatic sedimentary environments is critical to providing clarity on the factors that control microplastic deposition across the globe. Our previous work from Clayoquot Sound on the west coast of Vancouver Island, BC, Canada, suggested that regardless of sedimentary environment, MPs were ubiquitous, dominated by microfibers, and higher in concentrations in sediments with greater fractions of fine sediment. Here we examine the behavior of MPs in two sedimentary environments that are subject to different environmental and oceanographic conditions than those at Clayoquot Sound. First, we sampled the tidal mud flats in Boundary Bay, BC, a shallow bay bordering the urban and agricultural coastline of southwestern BC, Canada and the United States, with freshwater inputs from three local rivers and regional marine influences from the well-populated Salish Sea (Strait of Georgia and Puget Sound). We also sampled a core from Orchid Lake, a remote sub-alpine lake the in the Seymour Watershed within Metro Vancouver, BC, to better understand the controls on MP deposition in this remote sub-alpine region. While MPs have

been documented in high elevation lakes, many of these lakes, including Lake Hovsgol in Mongolia, lakes in the French Pyrenees, and sub-alpine lakes in Italy (Allen et al., 2019; Free et al., 2014; Sighicelli et al., 2018) are either in densely populated regions, or are surrounded by inhabitants or tourist camps (Velasco et al., 2020). Orchid Lake, with effectively no anthropogenic input, provides a unique opportunity to explore the role of atmospheric deposition of MPs in remote mountain catchments.

3.3. Methods

3.3.1. Study Area

Sediment cores were taken from two locations in Metro Vancouver: Boundary Bay and Orchid Lake (Figure 17).

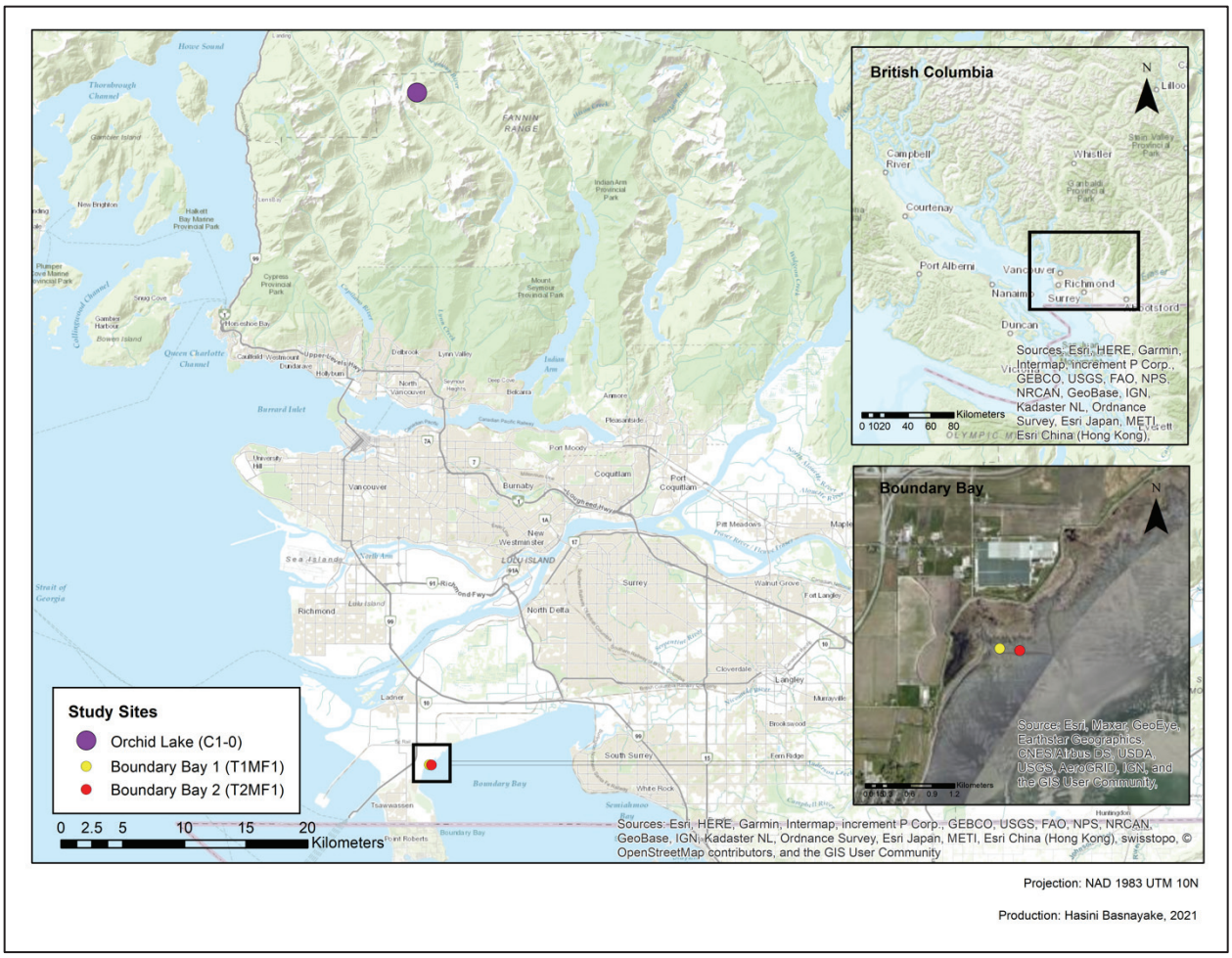


Figure 17. Map of study locations in Metro Vancouver including two sites at Boundary Bay (T1MF1 and T2MF1) in Metro Vancouver, and one site at Orchid Lake (C1-0) in the Seymour Watershed.

Boundary Bay

Two cores were collected from Boundary Bay, which is situated on the Canada-United States border between BC and Washington State (Figure 17). In Canada, Boundary Bay is adjacent to the densely populated region of Metro Vancouver in southwestern BC, with a population of 2.436 million. (Statistics Canada, 2016). The Canadian portion of Boundary Bay is protected by the Ministry of Forests, Lands, Natural Resource Operations and Rural Development (FLNRORD) as part of the Boundary Bay Wildlife Management Area.

Boundary Bay a 15 km long and 4 km wide headland-bay beach, part of the Fraser River Delta, with a total surface area of 61 km^2 (Shepperd, 1981). The main sources of freshwater to the Bay include the Serpentine and Nicomekl Rivers which flow into the northeast corner known as Mud Bay, the Campbell River flowing through Semiahmoo Nation, and two creeks on the US side, the California Creek and the Dakota Creek (Norman, 2013; Swinbanks & Murray, 1981). The cores collected for this analysis were collected from the low tidal flats. Boundary Bay is in the Strait of Georgia, and has a mixed semidiurnal tidal area, with two high tides and two low tides each day (Shepperd, 1981). Tides enter the Bay from the south, the flood tide being more concentrated on the east side and the ebb tide more concentrated on the west side (Baldwin & Lovvorn, 1994). Winds average 10-14 km/h and prevail from the east (Shepperd, 1981). The water in the Bay is clear, with salinity values similar to the Strait of Georgia at 24 to 29 (Shepperd, 1981). One potential source of MPs to Boundary Bay is contaminated water from the Strait of Georgia and Puget Sound, brought into Boundary Bay through tidal action (Frouin et al., 2013; Johannessen et al., 2015; Macdonald et al., 1991; Waldichuk, 1983).

Orchid Lake

Two cores were collected from Orchid Lake, located in the Seymour Watershed on the southwestern side of the Coastal Mountains just north of the City of North Vancouver. Orchid Lake is in a protected area that is closed to public access, therefore there is nearly no human interaction with the site. Surrounded by dense vegetation, the sub-alpine oligotrophic lake is at an elevation of 1030m, is 17m deep, and has a total area of 6 ha (Bonin, 2006). Orchid Lake is glacially fed, with a small creek at the southern end that discharges into the Seymour Watershed and eventually drains into the Lower Seymour Conservation area, the Burrard Inlet, and the Strait of Georgia. The Seymour Watershed has a storage capacity of 30 billion liters, and provides 40% of the water to the Greater Vancouver Regional District, and thus microplastics found in Orchid Lake may provide insight into potential contamination of drinking water (Bonin, 2006). The Coast Mountains experience high amounts of precipitation as storms moving over the Pacific Ocean release large amounts of precipitation when forced to rise over the North Shore Mountains or funnel into the deep dissected valleys. The region receives annual precipitation of 3000 mm (Bonin, 2006), which may result in significant wet deposition of microplastics from the atmosphere, as documented in other studies around the globe (Dris et al., 2016; Zhang et al., 2020) .

3.3.2. Field Methods

Two sediment cores were collected from Orchid Lake, and one of these cores was analyzed for the purpose of this study (Table 2). The site was accessed on September 20th and 21st in 2018 via helicopter with the assistance of Metro Vancouver personnel. The sediment cores were taken off of a zodiac boat using a gravity core with an internal piston that allows soft sediment to be captured with minimal to no compression or disturbance. A Secchi disk was first deployed to determine the depth of the sediment surface, which was 17 m. Samples were extracted from the cores in the field, at 1-cm intervals, and placed carefully into sterile sample bags. A core extractor

was used, and thus the subsamples did not come in contact with ambient air. The bags were kept in coolers until they were brought back to the laboratory and refrigerated at 4°C (Table 2).

Two sediment cores were collected in October 2018 in the low tidal flats of Boundary Bay in the western portion of the Bay. Cores were taken using a Livingstone Corer, which was pushed into the sediment until depth of refusal. No compression was observed within any of the cores. Cores were extracted in the field at 1-cm intervals into sterile sample bags and were handled carefully. The bags were kept in coolers until they were brought back to the laboratory and refrigerated at 4°C.

Table 2. Sediment core information for sampling sites at Orchid Lake and Boundary Bay.

Location	Core ID	Latitude	Longitude	Date Collected	Core length (cm)
Orchid Lake	C1-O	49.535914	-123.058266	18-09-2018	45.5
Boundary Bay	T1MF1	49.045615	-123.043105	13-10-2018	20
Boundary Bay	T2MF1	49.045465	-123.040689	14-10-2018	30

3.3.3. Laboratory Methods

Microplastic extraction, contamination control, and determination of sedimentary characteristics followed the methodology of Morra et al. (manuscript in preparation, Chapter 2), with some modifications.

In terms of microplastic extraction, the higher levels of organic carbon content in the Orchid Lake samples made visual identification of MPs more challenging and so we employed a slightly different methodology to remove organic material in these samples. First, 10 g of dried sample was added to a 1000-mL glass beaker. Then, 100 mL of 30% H_2O_2 was added to the beaker which

was covered with aluminum foil and left to sit for 24 hours. The sample was then transferred to 2 50-mL centrifuge tubes, using deionized water to ensure all material was transferred from the beaker to the tubes. The tubes were spun at 3000 rpm for 10 minutes. The supernatant from the centrifuge tubes was then decanted onto a Whatman50 (2.7µm pore size) filter paper in a Büchner funnel. The remaining sediment in the centrifuge tubes was subjected to 15 mL of 1.4 g/cm^3 sodium polytungstate per tube, resuspended and spun at 3000 rpm for 10 minutes. The supernatant was then pipetted onto the filter paper in a Büchner funnel. Samples were rinsed with DI five times to remove any excess hydrogen peroxide. The Büchner funnel was then quickly covered with aluminum foil to prevent MP contamination from ambient air. Filter papers were left to filter for 24 hours and were subsequently transferred to covered petri dishes for microscopic analysis.

Assessing contamination also followed the same methods found in Chapter 2. The daily contamination values (n=2) were averaged to produce an average contamination value unique to each core. Average daily contamination values ranged from 2 to 3 MPs. Across all sampling days, the average MP contamination value was 2 MPs (Appendix C). All data presented have been adjusted to account for average daily contamination values.

We analyzed the sediments at Boundary Bay and Orchid Lake for sediment properties that we hypothesized might influence MP deposition at the sites, including $\%C_{org}$ and sediment grain size distribution. Methodologies for these measurements also follow those outlined in Chapter 2.

3.3.4. Statistical Analyses

MP concentrations, $\%C_{org}$ and sediment grain size distribution were compared for significant differences between the three cores. All data were first tested for normality using the Shapiro-Wilk test for normality. As data were not normally distributed, the non-parametric Kruskal-Wallis test was used to test for significant differences between cores.

To test for differences in sedimentary environment influencing MP concentrations between the sites, the Wilcoxin Signed Rank Test for paired data was used to determine if there were any statistically significant differences in paired data between the sites. The paired data in this analysis included MP concentrations paired with $\%C_{org}$, the %silt and clay sediment fraction and the %medium sand sediment fraction. These data were graphed for visual representation and the Wilcoxin Signed Rank Test was used to determine if any differences between sites were statistically significant.

The significance level of all tests was set at $\alpha = 0.05$.

3.4. Results

3.4.1. Sediment Properties

The sediments at Orchid Lake were very fine, comprised predominantly of silt and clay (85%), with some very fine sand (10%) and the remainder fine sand (5%) (Figure 18). The sediments at Boundary Bay were coarser, represented predominantly by very fine sand (66%), followed by fine sand (19%), medium sand (11%), coarse sand (3%) and silt and clay (1%) (Figure 18) (Appendix G). Significant differences were observed between Orchid Lake and Boundary Bay for the distributions of the %silt and clay (Chi square=15.15 p=0.0005*, df=2), and the %medium sand grain size fractions (Chi square=14.723, p=0.0006*, df=2).

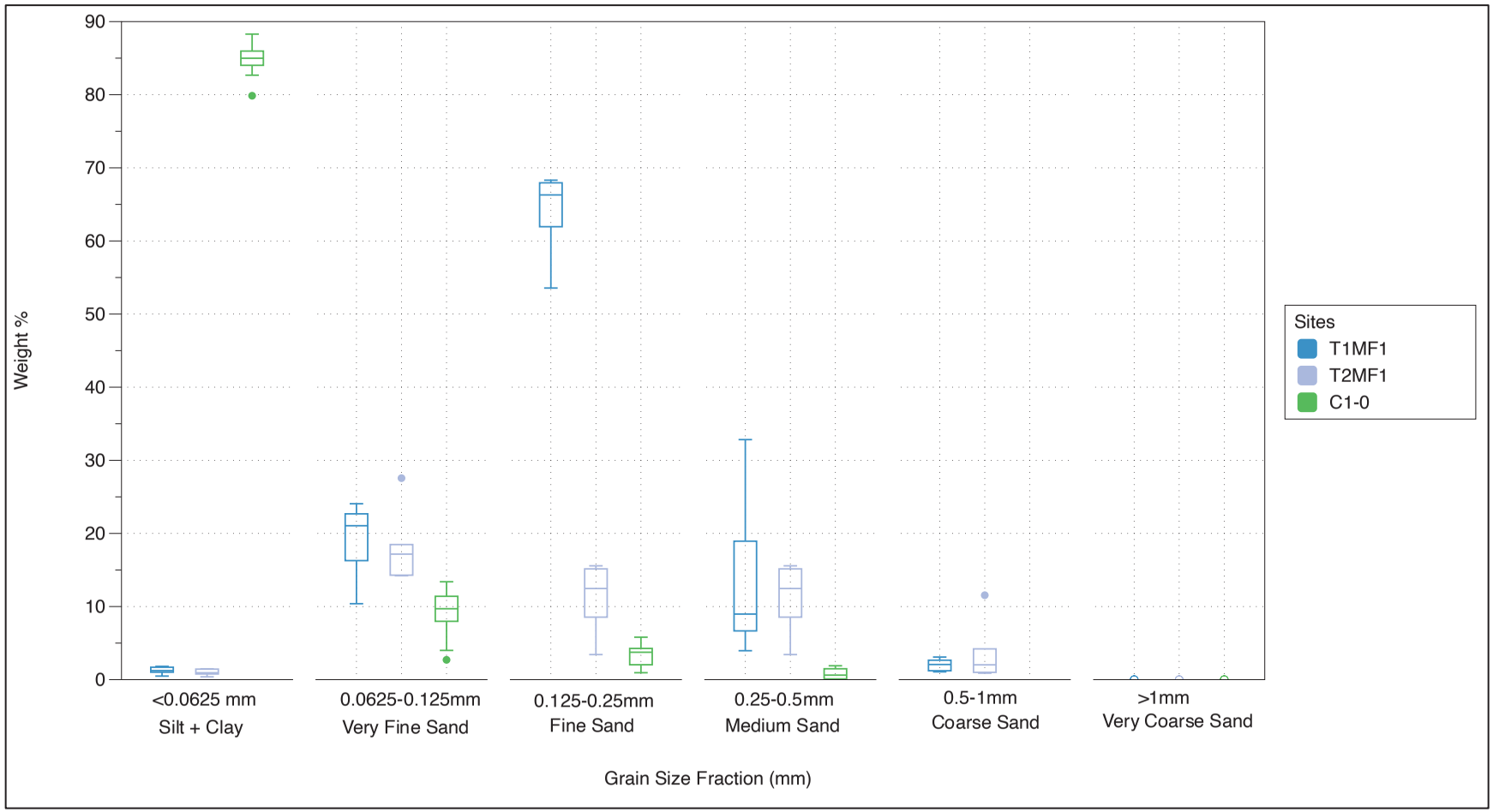


Figure 18. Box and whisker plots showing sediment grain size distribution at Boundary Bay (T1MF1 and T2MF1) and Orchid Lake (C1-0) for the entire lengths of the cores. Differences in grain size distribution for %silt and clay and %medium sand size fractions are statistically significant between Boundary Bay and Orchid Lake (%silt and clay: Chi square=15.15 p=0.0005*, df=2; %medium sand: 14.723, p=0.0006*, df=2).

To test the relationship between sediment grain size distribution and MP concentration between sites, we plotted the mean (+/- SE) %silt and clay fraction (Figure 19) and the mean (+/- SE) %medium sand fraction (Figure 20), which was the coarsest sediment observed at Orchid Lake, against mean (+/- SE) MP concentrations. While Orchid Lake, with the highest percentage of silt and clay size particles, had the highest MP concentrations, no significant differences were observed in paired data between sites for the %silt and clay or %medium grain size fractions and MP concentrations.

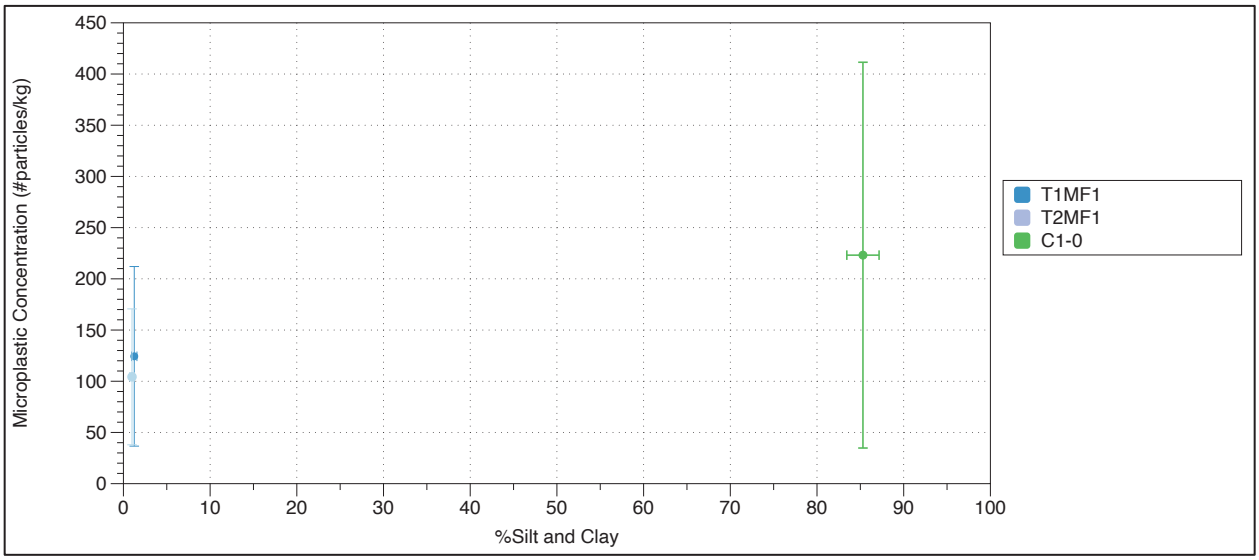


Figure 19. Mean (+/- SE) %silt and clay plotted against mean (+/- SE) MP concentration for the surface 10cm of core for Boundary Bay (T1MF1, T2MF1) and Orchid Lake (C1-0). Error bars indicate standard error. No significant differences observed in paired data between sites. (T1MF1:C1-0 (Z= 0.361, p=0.558), T2MF1:C1-0 (Z=0.600, p=0.452).

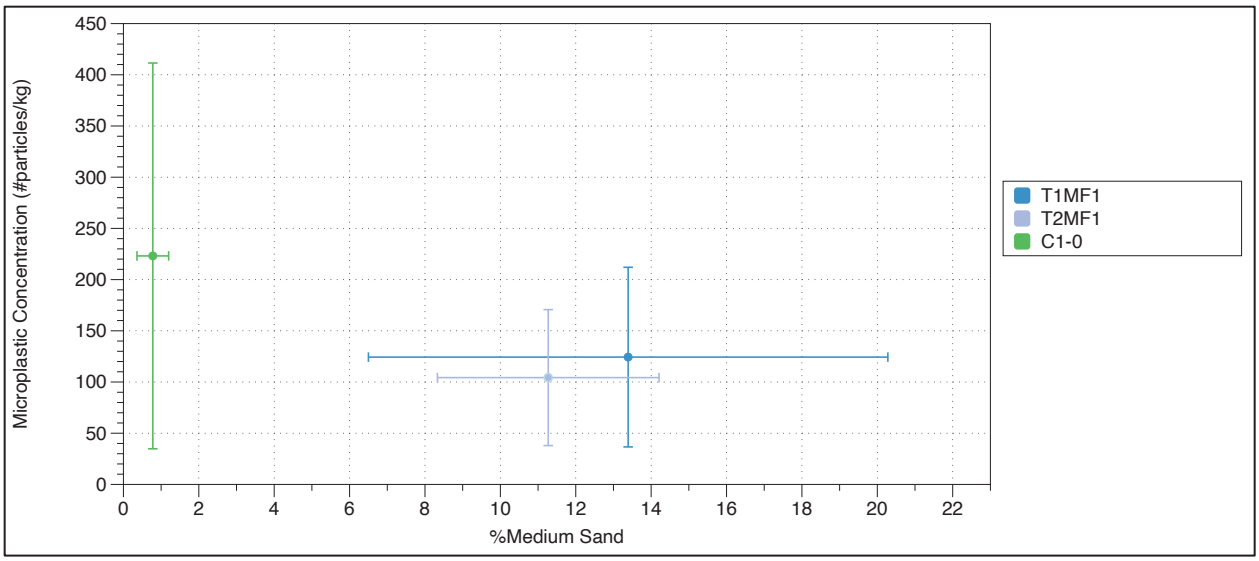


Figure 20. Mean (+/- SE) %medium sand plotted against mean (+/- SE) MP concentration for the surface 10cm of sediment for Boundary Bay (T1MF1, T2MF1) and Orchid Lake (C1-0). Error bars indicate standard error. No significant differences observed in paired data between sites. T1MF1:C1-0 (Z= 0.832, p=0.056), T2MF1:C1-0 (Z=0.732, p=0.985).

The $\%C_{org}$ at Orchid Lake ranged from 8.34-32.03% and was evenly distributed throughout the core, with no trends with depth observed (Appendix H). At Boundary Bay, $\%C_{org}$ was extremely low and ranged from 0-0.77%. Similar to Orchid Lake, no trends at depth were observed in $\%C_{org}$ within the Boundary Bay cores. While differences in $\%C_{org}$ were statistically significant between sites (Chi square = 19.685, p=0.001*, df=2), no significant differences were observed between the paired comparisons of mean $\%C_{org}$ and mean MP concentration (Figure 21).

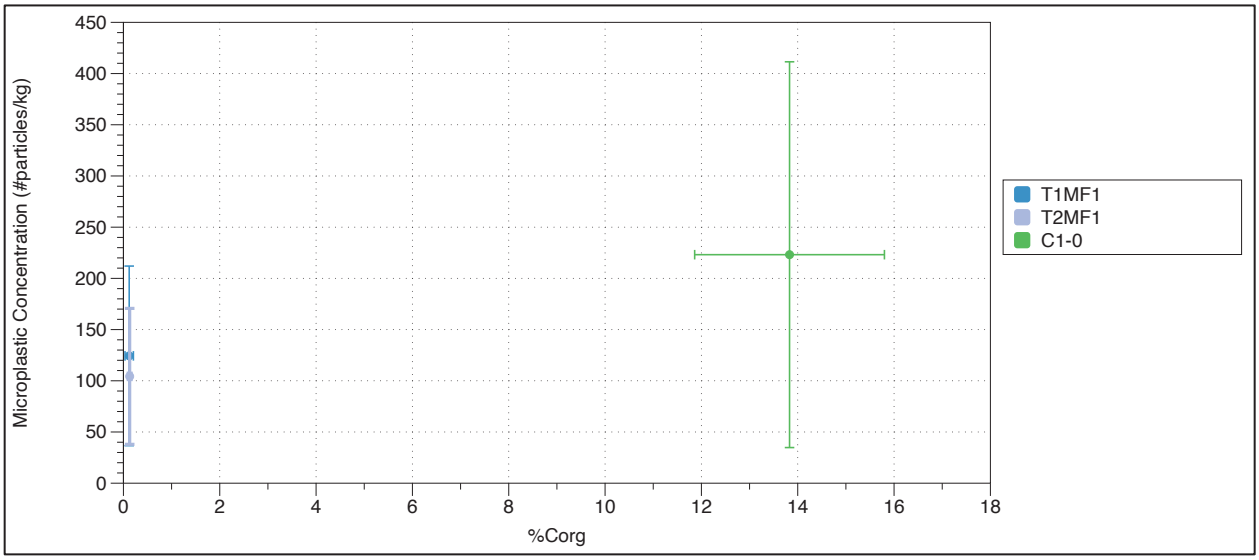


Figure 21. Mean (+/- SE) %Corg plotted against mean (+/- SE) MP concentration for Boundary Bay (T1MF1, T2MF1) and Orchid Lake (C1-0). Error bars indicate standard error. No significant differences observed in paired data between sites. T1MF1:C1-0 (Z= 0.089, p=0.767), T2MF1:C1-0 (Z=0.303, p=0.586), GB REF:RP REF (Z=0.099, p=0.756).

3.4.2. Microplastic Characteristics

Microfibers were the sole type of plastic particle found in all samples at Boundary Bay and Orchid Lake, and appeared as thin, weathered strands. Microfibers were significantly longer at Boundary Bay than at Orchid Lake and ranged in size from 0.49-4.10 mm, whereas microfibers at Orchid Lake ranged from 0.24-0.87 mm (Figure 22). Microfibers at Orchid Lake were significantly smaller than at Boundary Bay (Chi square = 53.23, p= 0.0001, df=2).

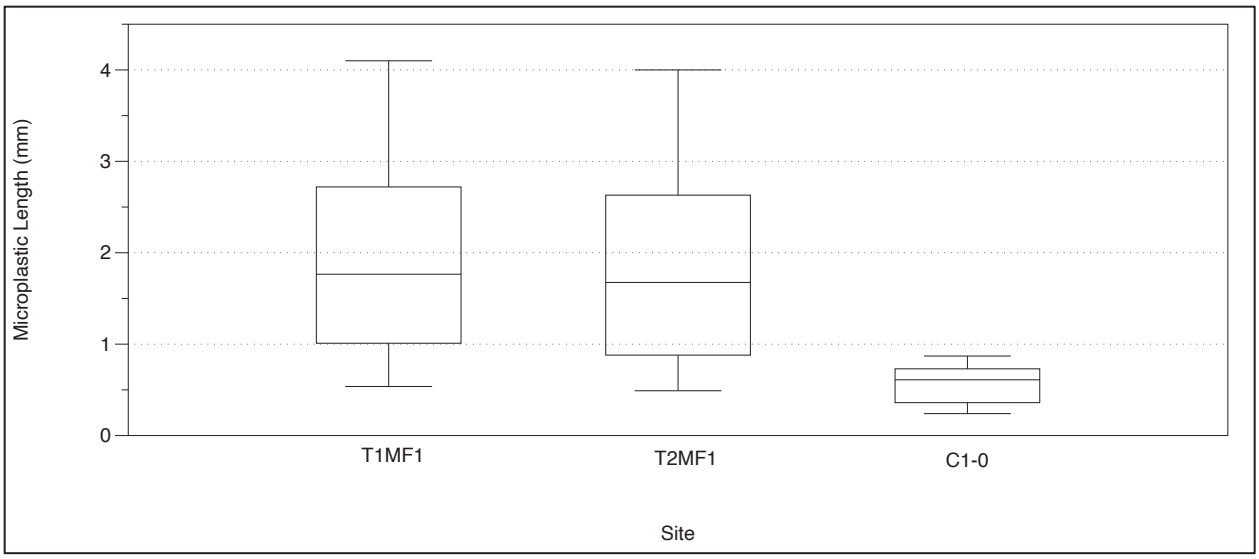


Figure 22. Box and whisker plots showing microplastic length (mm) at Boundary Bay (T1MF1 + T2MF1) and Orchid Lake (C1-0) (Chi square= 53.23, p= 0.0001, df=2).

Interestingly, the MPs at Boundary Bay displayed a similar range of colours to those found at Orchid Lake, predominantly black and blue. MP colour distribution at Boundary Bay was 55% black, 38% blue, 4% grey, 2% red and 1% green, while MP colour distribution at Orchid Lake was 63% black 27% blue, and 10% grey (Figure 23).

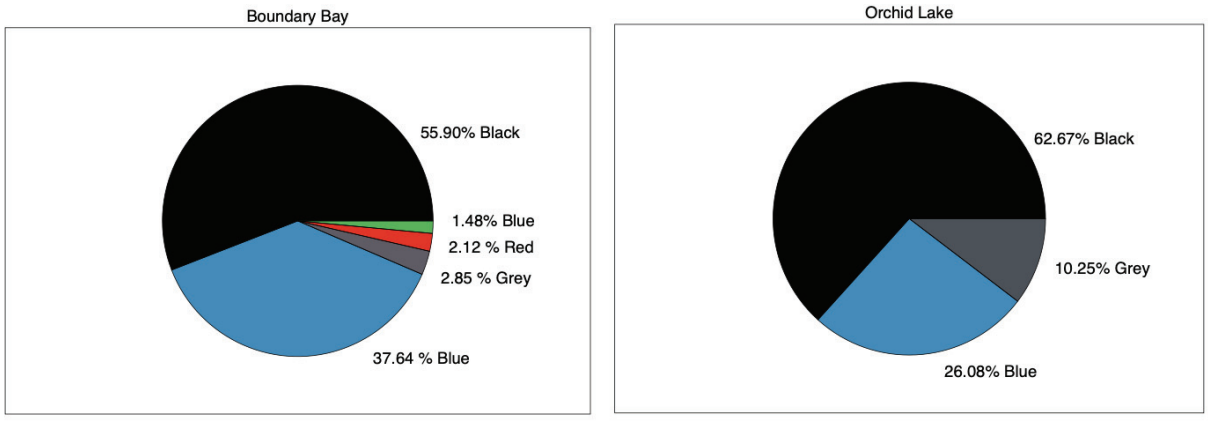


Figure 23. Microplastic colour distribution at Boundary Bay (T1MF1+T2MF1) and Orchid Lake (C1-0).

3.4.3. Microplastic Concentrations

Across all sites, MP concentrations were highest in the surface layers, and showed a decreasing trend at depth (Figure 24). MP concentrations at Boundary Bay and Orchid Lake were found predominantly in the top 10 cm of the core. We chose a depth horizon of 10 cm for calculations of mean and median MP concentrations at the sites to be representative of when MPs are seen to be deposited in the sediments. MPs were not evenly distributed throughout the core and showed a high degree of variability at depth (Figure 24). The large number of 0 values can be attributed to data correction for contamination, while sedimentary processes likely explain variability at depth. At Boundary Bay, the high energy events and significant bioturbation documented at the site by Dashtgard (2011) could explain this variability at depth. Orchid Lake represents a very low energy environment and variability of MPs at depth is more difficult to diagnose in these sediments. It is possible that seasonal variations in precipitation and snowmelt are causing this variability.

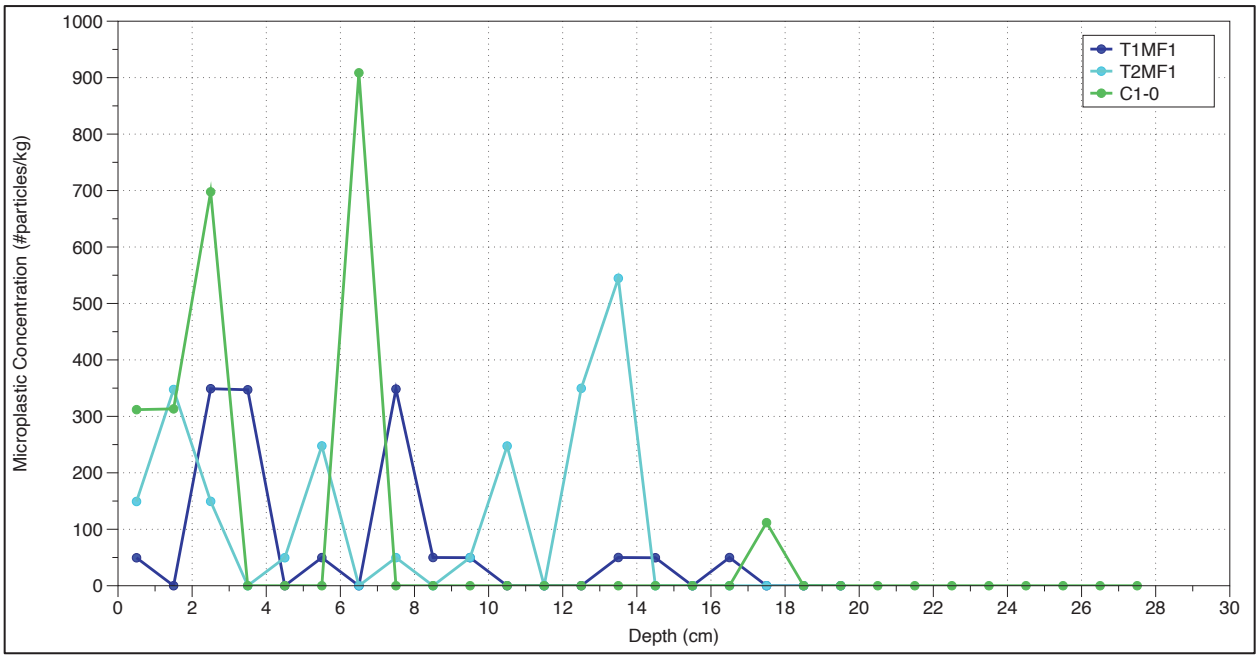


Figure 24. Microplastic concentration (#particles/kg) at depth for Boundary Bay (T1MF1, T2MF1) and Orchid Lake (C1-0). All data corrected for contamination.

At Boundary Bay, MP concentrations ranged from 0-349 particles/kg with a mean (+/- SE) and median concentration of 114 ± 61 particles/kg, and 50 particles/kg, respectively. At Orchid Lake, MP concentrations ranged from 0-908 particles/kg with a mean (+/- SE) concentration of 223 ± 188 particles/kg (Appendix E). Differences in MP concentration between the sites were not statistically significant.

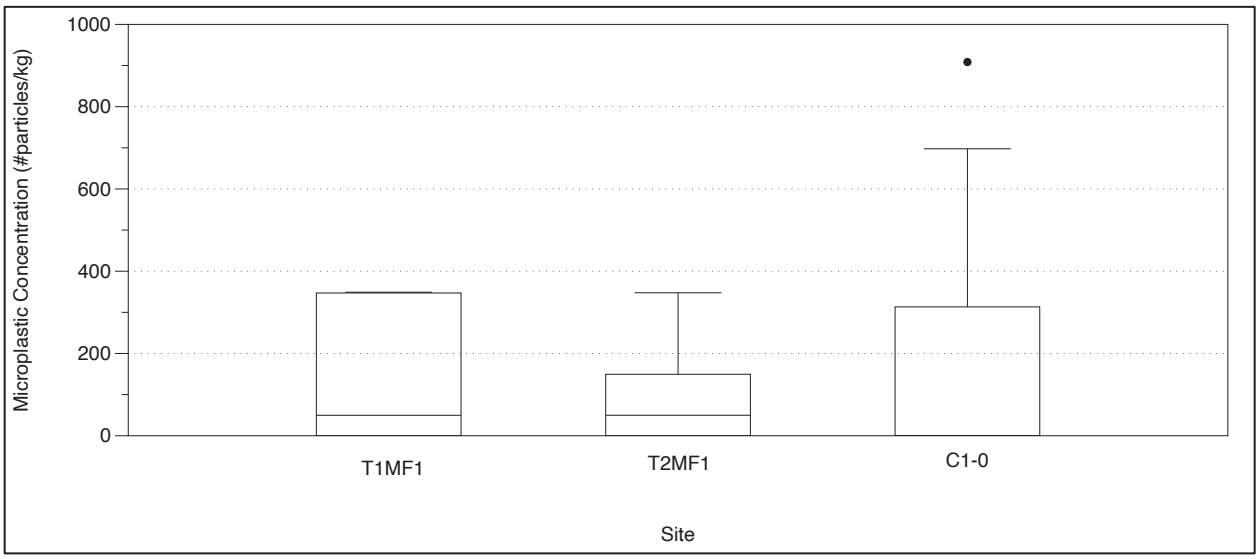


Figure 25. Box and whisker plot showing MP concentration (#particles/kg) at Orchid Lake (C1-O) and Boundary Bay (T1MF1 and T2MF1) over the surface 10cm of core. No significant differences observed between sites. (Chi square = 0.529, p=0.767, df=2).

3.5. Discussion

3.5.1. Ubiquity of Microplastics

Our findings from Boundary Bay and Orchid Lake support the literature documenting the ubiquity of MPs in aquatic sedimentary environments (Ajith et al., 2020; Boucher & Friot, 2017; Gago et al., 2017). Orchid Lake was sampled to serve as a control for the region since it is part of a closed system with effectively no anthropogenic input. The Orchid Lake core was exposed to less

contamination in the field, as the core was not exposed to ambient air before being extruded and subsampled at 1-cm intervals. The MP concentrations we determined were also corrected for contamination, so it is unlikely that contamination is responsible for the high concentrations of MPs in the surface sediments at Orchid Lake. This study is not the first to document MPs in remote high-elevation regions. MPs have been documented in Lake Hovsgol, Mongolia, in the Tibetan Plateau, and in the French Pyrenees (Allen et al., 2019; Feng et al., 2020; Free et al., 2014). These findings support the notion that MPs are pervasive and are found in all environmental media in which they are analyzed for, regardless of the remoteness of the location (Andrady, 2011; Hidalgo-Ruz et al., 2013; Wright et al., 2013).

3.5.2. Relationship Between Microplastics and Depositional Environment

Boundary Bay and Orchid Lake represent two very different aquatic sedimentary environments, and this is likely to play a large role in the distribution of MPs between the sites, as well as within the respective cores. Orchid Lake represents a relatively low energy environment, with MPs entering the lake through aerial deposition via precipitation or snowmelt, with some MPs settling out of the water column and into the sediment. As an oligotrophic sub-alpine lake, there is less activity happening at the sediment-water interface, and within the core itself. This is contrasted with Boundary Bay, which exhibits sediment remobilization as a result of strong tidal action and storms, which may redistribute MPs and/or carry MPs out of the area (Dashtgard, 2011). Dashtgard (2011) also showed that Boundary Bay has high sediment disruption values from burrowing organisms that redistribute sediment. These factors likely explain why Orchid Lake exhibits higher MP concentrations than Boundary Bay.

Studying the sedimentary properties at Boundary Bay and Orchid Lake, including sediment grain size distribution and $\%C_{org}$ provided further clarity on the sedimentary environment at the sites. The sediments at Boundary Bay and Orchid Lake were analyzed for sediment grain size

distribution and $\%C_{org}$ to determine if these factors influence MP deposition at the sites. Orchid Lake had the highest $\%C_{org}$ and the finest sediments (85% silt + clay), indicative of a very low energy environment. The sediments at Boundary Bay were coarser, consisting predominantly of fine sand, indicative of a higher energy environment, but Boundary Bay also showed very low $\%C_{org}$. While Boundary Bay and Orchid Lake had significantly different sediment grain size distributions for the %silt and clay and the %medium size fractions, as well as significantly different $\%C_{org}$, when paired with MP concentrations, no significant differences were observed between sites. The lack of a relationship between the sedimentary properties and MP concentrations between the sites may be the result of the sites representing very different sedimentary environments with one site part of a closed freshwater system in the sub-alpine, and the other site an open marine system in an urban environment.

Although the cores at Orchid Lake and Boundary Bay were not dated, the strong relationship between MP concentration and depth documented in Clayoquot Sound can also be observed at these sites. What we see in the cores from Boundary Bay and Orchid Lake, similarly to Clayoquot Sound, is a distinct plastic horizon in the sediments above which MPs are present, regardless of the depositional environment in which they are found. We see this plastic horizon at 10 cm depth at Orchid Lake and Boundary Bay. This plastic horizon may be indicative of the proposed epoch known as the Anthropocene, characterized by the onset of industrialization, mass environmental degradation, plastic proliferation, and other environmental stressors (Irabien et al., 2020; Lewis & Maslin, 2015).

3.5.3. Sources of Microplastics

Deposition of MPs in the sediments of Boundary Bay is likely controlled by a combination of factors including the abundance of MPs in the local marine environment and proximity to urban centres. In the surface waters of the Strait of Georgia, tidal currents are strongly influenced by the

Fraser River (LeBlond, 1983). The Fraser River flows through densely populated regions of Metro Vancouver, and is known to contain a number of contaminants, such as hydrocarbons and heavy metals (Johannessen & Ross, 2002). While there have not been studies carried out on MPs in the Fraser, it is likely that the water in the Fraser River is highly contaminated with MPs. While the Fraser River does not have a direct effect on Boundary Bay (Dashtgard, 2011), water from the Fraser River that enters the Strait of Georgia is subject to various oceanographic controls in the Strait, some of which could result in MPs making their way to Boundary Bay. Studies on marine pollution in the region show that local environmental degradation from waters on the periphery of the Strait of Georgia and Puget Sound, adjacent to urban communities and industries, has resulted in an influx of pollutants into the Strait (Macdonald et al., 1991). These pollutants include wastewater effluent from a number of wastewater treatment facilities (Johannessen & Ross, 2002; Johannessen et al., 2015; Macdonald et al., 1991; Yunker et al., 1999). Wastewater has been shown to be a major source of MPs to the marine environment, and one wastewater treatment plant in Vancouver alone is known to emit 30 billion tonnes of MPs into the receiving environment every year (Gies et al., 2018). These contaminated waters from the Strait of Georgia and Puget Sound make their way to the Bay through tidal currents which flow upward from the south, following the coast of Washington State and turning toward Vancouver Island (Waldichuk, 1957),

Desforges et al. (2014) documented the presence of MPs in the subsurface seawater of the Strait of Georgia in the waters adjacent to Boundary Bay and found concentrations of MPs to be 12-fold greater in these waters than in the offshore Pacific Ocean. The study also found that the predominant MPs in the region, microfibers, were more concentrated in the nearshore waters adjacent to the lower mainland, likely as a result of anthropogenic input (Desforges et al., 2014). This water contaminated with MPs is brought into the Bay through tidal action and thus we believe

that the abundance of MPs in the subsurface seawater in the region plays a large role in the deposition of MPs in Boundary Bay.

Boundary Bay has two rivers that flow into it from the northeast, the Serpentine and the Nicomekl, as well as the Little Campbell River, which flows through highly urbanized areas before draining into Boundary Bay. There are also two small creeks, the Dakota and California Creek that drain into the Bay. These rivers may carry MPs into the Bay and could be responsible for some of the deposition at the study sites, but a study by Dashtgard (2011) documented the limited influence of these freshwater sources on sedimentation at Boundary Bay, and thus these rivers are not likely to be significant sources of MPs to the study sites. Additionally, the majority of MPs would likely be trapped by the dense vegetation and organic matter in the marsh before reaching the intertidal zone as salt marshes have been shown to effectively capture MPs (Lloret et al., 2021).

3.5.4. The Significance of Orchid Lake

We hypothesized that MPs in the sediments of Orchid Lake are the result of atmospheric deposition. Allen et al. (2019) documented wet and dry deposition of MPs from the atmosphere in a remote, pristine catchment in the French Pyrenees Mountains at rates as high as 44 fibers/m^2/day. Other work in remote mountain catchments in Tibet, as well as in the urban environment in Paris, have suggested that precipitation events lead to substantial wet deposition of MPs from the atmosphere (Allen et al., 2019; Dris et al., 2016; Feng et al., 2020). Dris et al. observed atmospheric fallout ranging from 11-355 particles/m^2/day during rainy periods (from 2-5 mm/day). While this study was carried out in an urban location in Paris, it provides substantive evidence for what might be causing MP deposition at Orchid Lake. Orchid Lake receives annual precipitation of approximately 3000 mm, which based on the findings of Dris et al. (2019) could result in high wet deposition rates of MPs. Studies documenting atmospheric transport of MPs in the Pyrenees and Tibet documented MPs of a similar size range to Orchid Lake (Allen et al.,

2019; Feng et al., 2020). In the Pyrenees, microfibers were documented ranging in size from 0.05-0.75 mm, while in the Tibetan Plateau the microfibers ranged from 0.05-0.5 mm (Allen et al., 2019; Feng et al., 2020). Similarly, MPs at Orchid Lake ranged in size from 0.24 to 0.81 mm and were significantly smaller than the MPs at Boundary Bay. The significantly smaller MPs documented at Orchid Lake compared to Boundary Bay, which are similar in size to MPs from other remote regions, may suggest that these particles are deposited via long-range atmospheric transport. Studies on atmospheric transport of MPs have noted that smaller MPs are more likely to travel via long-range atmospheric transport, and break down to into smaller particles as a result of mechanical abrasion and chemical weathering in the atmospheric environment (Zhang et al., 2020). Microfibers from aerial deposition of MPs in urban environments in Hamburg and Paris were found to be several orders of magnitude larger than in remote regions (Zhang et al., 2020). Orchid Lake presents a unique opportunity to study the influence of atmospheric transport in remote regions, and may point to a baseline of atmospheric deposition from long-range transport in the region.

3.6. Conclusion

Characterizing MP concentrations in different aquatic sedimentary environments will provide further clarity on their ubiquity, persistence in the environment, and potential impacts to aquatic organisms and environments. In Chapter 3, MP concentrations in two different aquatic sedimentary environments, an urban bay, and a remote sub-alpine lake, were compared to test the relationship between sedimentary environment and location, urban vs. remote, on MP concentration. The results of this research solidify the ubiquity of MPs in aquatic sedimentary environments, regardless of location, with MPs detected in all sediment cores. Surprisingly, the MP concentrations at Orchid Lake were higher than those observed at Boundary Bay, which may be a result of the high energy environment at Boundary Bay reworking sediments and MPs. No relationships were detected between sediment grain size or percent organic carbon ($\%C_{org}$) and

MP concentrations between the sites, but as the sites represent very different sedimentary environments, these comparisons may be difficult to comment on accurately. The likely sources of MPs to Orchid Lake are from aerial deposition, which has been noted in other remote alpine and sub-alpine lakes (Allen et al., 2019; Free et al., 2014; Velasco et al., 2020). At Boundary Bay, proximity to wastewater effluent and contaminated waters from the Fraser River, Strait of Georgia, and Puget Sound are likely sources of MPs, with tidal currents bringing contaminated water into the Bay. Microfibers were the sole type of MP particle found, which corroborates other research that puts high quantities of microfibers in the waters of the Strait of Georgia and Puget Sound (Desforges et al., 2014).

Chapter 4. General Conclusion

In this study we quantified MP concentrations and MPARs in different aquatic sedimentary environments in southwestern BC and discussed the potential controls on MP deposition and distribution in the sediments. We found MPs to be ubiquitous across all study sites, regardless of aquatic sedimentary environment or whether the location was urban or remote. We also looked at sediment properties, namely sediment grain size distribution and $\%C_{org}$ to better understand the sedimentary environments at the sites.

Mean (+/- SE) MP concentrations plotted against mean (+/- SE) $\%C_{org}$ across all sites is depicted in Figure 26. Orchid Lake had the highest $\%C_{org}$, indicative of a low energy environment (Enders et al., 2019), while the marine sites all showed very low $\%C_{org}$ values (<2.9%), indicative of higher energy environments. While there were significant differences in $\%C_{org}$ observed between the marine sites, these low $\%C_{org}$ values made it difficult to discern differences in sedimentary environment using this parameter (Figure 26). Although Orchid Lake had significantly higher $\%C_{org}$ values, when paired with MP concentrations, no significant differences were observed between any of the sites.

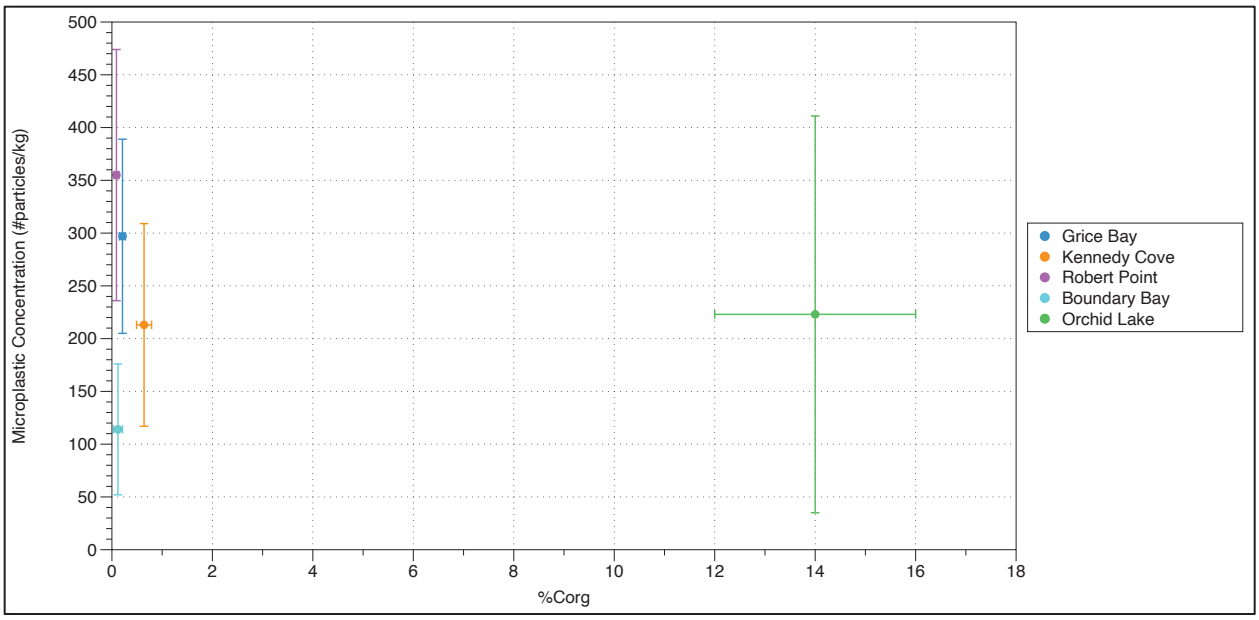


Figure 26. Mean microplastic concentration (#particles/kg) plotted against mean %Corg for the surface 10cm of core at all sites including Grice Bay (GB REF), Kennedy Cove (KC 3), Robert Point (RP REF), Boundary Bay (T1MF1 + T2MF1) and Orchid Lake (C1-0). Error bars indicate standard error. No significant relationships were observed between microplastic concentrations and %Corg between sites.

Mean (+/- SE) sediment grain size distribution for the %fine fraction (very fine sand, fine sand, and silt + clay) and mean (+/- SE) sediment grain size distribution for the % coarse fraction (medium sand, coarse sand, and very coarse sand) were plotted against mean (+/- SE) MP concentrations at all sites (Figures 27 and 28, respectively). Sediment grain size distribution at the marine sites indicated that Boundary Bay had a similar, intermediate, energetic environment to Grice Bay, whereas Kennedy Cove had much coarser sediments, indicative of a high energy environment (Figure 28), and Robert Point more fine sediments, indicative of a low energy environment (Figure 27). When comparing mean %fines against mean MP concentrations across all sites (Figure 27), a significant difference was observed between Robert Point and Kennedy Cove ($Z=3.381$, $p=0.003$). This finding suggests that MPs may tend to be deposited in finer sediment in lower energy environments, as has been documented elsewhere (Courtene-Jones et al., 2020; Maes et al., 2017; Vianello et al., 2013). No significant differences were observed in

paired data between the %coarse sediment grain size fractions and MP concentrations at any of the sites (Figure 28).

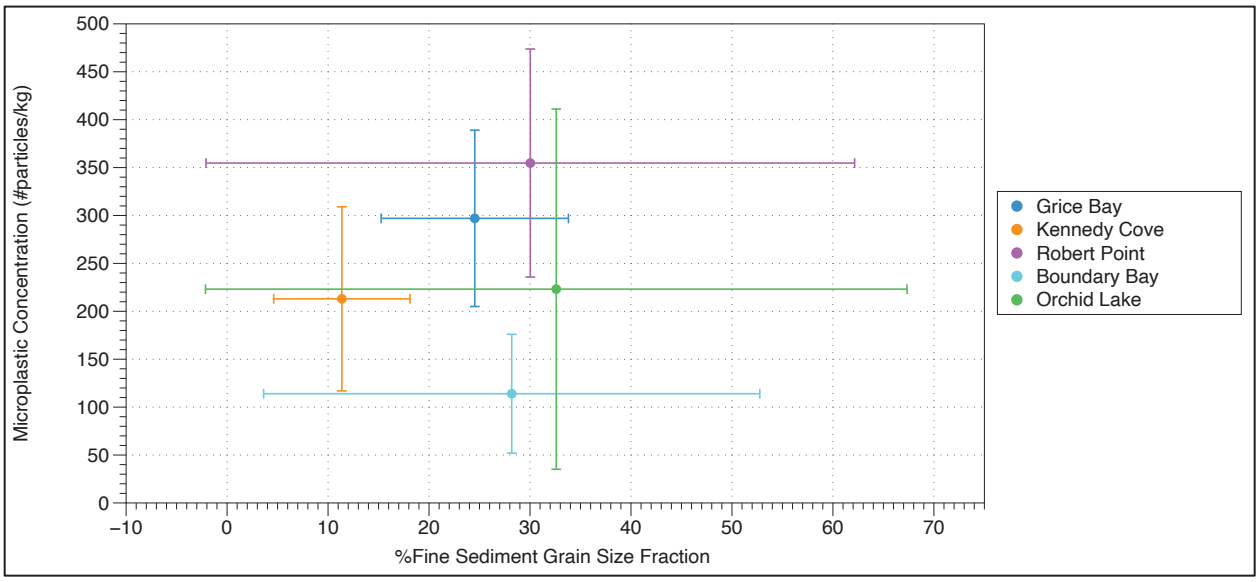


Figure 27. Mean microplastic concentrations (#particles/kg) plotted against the mean %fine sediment grain size fraction (%fine encompassing fine sand, very fine sand, and silt + clay) for the surface 10cm of sediment. Error bars indicate standard error. Significant differences were detected between Robert Point and Kennedy Cove (Z=3.381, p=0.003).

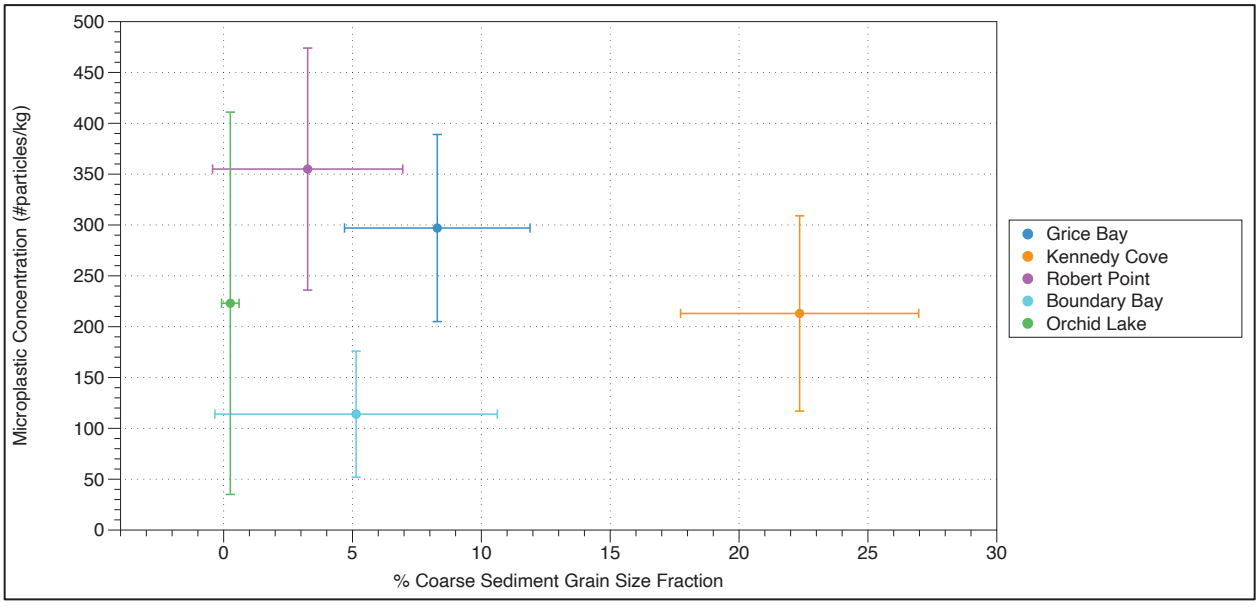


Figure 28. Mean microplastic concentrations (#particles/kg) plotted against the mean %coarse sediment grain size fractions at all sites including Grice Bay (GB REF), Kennedy Cove (KC3), Robert Point (RP REF), Boundary Bay (T1MF1 + T2MF1) and Orchid Lake (C1-0). Error bars indicate standard error. No significant differences observed in paired data between sites.

Mean (+/- SE) microplastic concentrations observed in Clayoquot Sound were 288 ± 92 particles/kg, approximately 2.5x greater than Boundary Bay (114 ± 61 particles/kg) and 1.3x greater than Orchid Lake (223 ± 188 particles/kg). MPs were not evenly distributed throughout the cores and were highly variable at depth (Figure 29). Normalization for contamination likely explains many of the 0 values in our data, while sedimentary processes may explain the patchy distribution of MPs. Oceanographic conditions such as strong tidal action, high energy events, and upwelling that are common at the marine sites (Okey & Dallimore, 2015; Dashtgard, 2011) may partially explain the variable vertical distribution of MPs. Bioturbation is another factor that has been shown to redistribute sediments in Boundary Bay and might play a role in the redistribution of MPs in the sediment column (Dashtgard, 2011). At the Orchid Lake site, causes for variability are more difficult to discern, although they may relate to seasonal variation in precipitation and snowmelt.

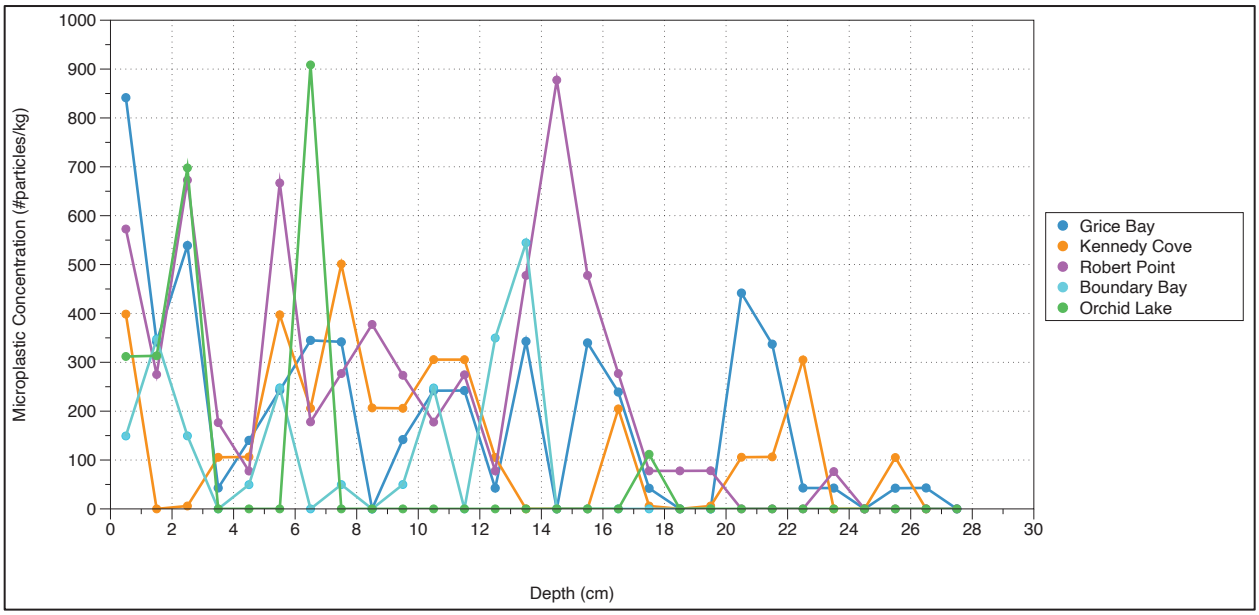


Figure 29. Microplastic concentrations (# particles/kg) at depth across all sites including Grice Bay (GB REF), Kennedy Cove (KC3), Robert Point (RP REF), Boundary Bay (T1MF1 + T2MF1) and Orchid Lake (C1-0).

We also compared mean MP concentrations and mean MPARs from our sites to sites around the world and found that our MP concentrations were of a similar magnitude to the majority of global sites with mean concentrations ranging from 0-500 particles/kg. The mean MPARs documented at Clayoquot Sound over the last ~30 years (14 ± 5 particles/100cm^2/year) were also of a similar order of magnitude to all studies that documented MPARs over the same time period in marine sedimentary environments, ranging from 2-35 particles/100cm^2/year.

Oceanographic conditions in the region likely explain why MP concentrations are higher in Clayoquot Sound than Boundary Bay, as contaminated water from the Strait of Georgia travels towards Vancouver Island in the VICC bringing with it contaminated water from the Fraser River (Freeland et al., 1984). In Clayoquot Sound, water from the northeastern Pacific travelling in the Alaskan and California Currents also bring contaminated water to the sites (Freeland et al., 1984). Boundary Bay is largely protected from the Fraser River (Dashtgard, 2011), although it is influenced by heavily contaminated water from the Strait of Georgia and Puget Sound, with tidal currents likely bringing contaminated water from these regions into the Bay (Shepperd, 1981). High-energy events that are common at Boundary Bay, which rework sediments and thus MPs, may be responsible for the lower concentrations of MPs at Boundary Bay compared to the other sites (Dashtgard 2011). At Clayoquot Sound and Boundary Bay, wastewater effluent is believed to be a major source of MPs to the study sites. Additionally, at Clayoquot Sound, aquaculture activities near the sites are likely to contribute to MP deposition.

At Orchid Lake, a remote sub-alpine site with no anthropogenic input, we believe aerial deposition is responsible for MP deposition, as this has been shown to be a mechanism of transport in other remote mountain regions (Allen et al., 2019; Feng et al., 2020; Free et al., 2014).

Mean (+/- SE) MPARs in the region increased from 7 ± 3 particles/100cm^2/year in 1950 to 33 ± 12 particles/100cm^2/year in 2016, with increases mirroring rates of global of plastic production

and increases in aquaculture tenures in the region. This temporal analysis also revealed a plastic horizon in the sediments in the 1950s that may denote the onset of the Anthropocene (Lewis & Maslin, 2015). Future studies on MPs in aquatic sedimentary environments in BC and globally will need to move beyond simply studying MP concentrations and focus on determining MPARs in the sediments in order to better understand relationships between MPs and sedimentary processes as well as source dynamics. Considering MPs have only been in the environment for no more than 80 years, ^{210}Pb dating is an effective dating option as it can accurately date sediments as old as 100 years of age (Barsanti et al., 2020). In terms of the various models based on ^{210}Pb dating, the CRS model allows for sedimentation rates to vary over time and may provide more robust chronologies (Turner & Delorme, 1996; Uddin et al., 2021). Another more cost-effective option for dating sediment cores is the use of the ^{137}Cs marker which provides a marker for the 1963 horizon of maximum weapons testing (Uddin et al., 2021). While this method would provide some indication of how MPs behave at depth, the lack of chronological resolution would make it difficult to make inferences about potential sources of MPs, and thus ^{137}Cs is better used to assess the efficacy of the ^{210}Pb model.

The Orchid Lake core provides an interesting opportunity to study atmospheric deposition of MPs in a drinking watershed. In order to further assess aerial deposition of MP contamination in a system like the Seymour Watershed, it would be effective to sample various locations including the snowpack, source lakes, sediment, and rivers, as well as the reservoirs before and after filtration. This information would provide a more well-rounded picture of how MPs are behaving in the watershed and whether or not they are being effectively removed from the drinking water supply. Additional clarity will be needed on the toxicological impacts of MPs to provide greater context on the potential impacts of MPs to various species and ecosystems. Understanding thresholds for harm in humans will be important to determine whether or not studies on MPs in drinking watersheds are necessary.

One major limitation to our study was the lack of dated sediment cores at Orchid Lake and Boundary Bay, which did not allow for us to observe temporal trends in MPs at depth or calculate MPARs for these sites.

Another limitation to our study was the lack of identification of polymers by Fourier transform infrared spectroscopy (FTIR) (Veerasingam et al., 2020). FTIR analysis provides information on the specific polymer types found in the sediments. Having information on polymer types is a key piece of the puzzle when it comes to trying to discern the possible sources and fate of MPs in the aquatic environment. While this is an important piece, FTIR analysis of MPs is not a perfect science and does not always provide clear information on polymer identity when analyzing MPs that have experienced physical and/or chemical weathering (Xu et al., 2019). Spectral changes that occur during these processes are poorly understood (Xu et al., 2019). Furthermore, FTIR analysis requires one to have a very comprehensive spectral library and robust matching algorithm to ensure accuracy (Xu et al., 2019).

Despite these limitations, this work highlighted a number of key messages regarding MP deposition in aquatic sedimentary environments. Our findings support the ubiquity of MPs, regardless of environment type or whether the location is urban or remote. Our results also indicate that MPs may preferentially be deposited in finer sediments, associated with lower energy environments. Waste-water effluent and aquaculture are likely major contributors to MP deposition in aquatic sedimentary environments. Our findings from Orchid Lake point to a baseline of deposition from long-range atmospheric transport in the region. This study also contributes to the growing body of science observing temporal trends in MP deposition in aquatic sedimentary environments, which will be a key piece of the puzzle to identifying relationships between MP deposition, source dynamics, and sedimentary processes.

References

Abidli, S., Antunes, J. C., Ferreira, J. L., Lahbib, Y., Sobral, P., & Trigui El Menif, N. (2018). Microplastics in sediments from the littoral zone of the north Tunisian coast (Mediterranean Sea). *Estuarine, Coastal and Shelf Science*, *205*, 1–9. https://doi.org/10.1016/j.ecss.2018.03.006

Abril, J. M. (2019). Radiometric dating of recent sediments: On the performance of 210Pb-based CRS chronologies under varying rates of supply. *Quaternary Geochronology*, *51*(December 2018), 1–14. https://doi.org/10.1016/j.quageo.2018.12.003

Ajith, N., Arumugam, S., Parthasarathy, S., Manupoori, S., & Janakiraman, S. (2020). Global distribution of microplastics and its impact on marine environment—a review. In *Environmental Science and Pollution Research*. Springer. https://doi.org/10.1007/s11356-020-09015-5

Akdogan, Z., & Guven, B. (2019). Microplastics in the environment: A critical review of current understanding and identification of future research needs. *Environmental Pollution*, *254*, 113011. https://doi.org/10.1016/j.envpol.2019.113011

Akhbarizadeh, R., Moore, F., Keshavarzi, B., & Moeinpour, A. (2017). Microplastics and potentially toxic elements in coastal sediments of Iran's main oil terminal (Khark Island). *Environmental Pollution*, *220*, 720–731. https://doi.org/10.1016/j.envpol.2016.10.038

Al-Lihaibi, S., Al-Mehmadi, A., Alarif, W. M., Bawakid, N. O., Kallenborn, R., & Ali, A. M. (2019). Microplastics in sediments and fish from the Red Sea coast at Jeddah (Saudi Arabia). *Environmental Chemistry*, *16*(8), 641–650. https://doi.org/10.1071/EN19113

Alimba, C. G., & Faggio, C. (2019). Microplastics in the marine environment: Current trends in environmental pollution and mechanisms of toxicological profile. *Environmental Toxicology and Pharmacology*, *68*(March), 61–74. https://doi.org/10.1016/j.etap.2019.03.001

Allen, S., Allen, D., Phoenix, V. R., Le Roux, G., Durántez Jiménez, P., Simonneau, A., Binet, S., & Galop, D. (2019). Atmospheric transport and deposition of microplastics in a remote mountain catchment. In *Nature Geoscience* (Vol. 12, Issue 5, pp. 339–344). https://doi.org/10.1038/s41561-019-0335-5

Alomar, C., Estarellas, F., & Deudero, S. (2016). Microplastics in the Mediterranean Sea: Deposition in coastal shallow sediments, spatial variation and preferential grain size. *Marine Environmental Research*, *115*, 1–10. https://doi.org/10.1016/j.marenvres.2016.01.005

Alves, V. E. N., & Figueiredo, G. M. (2019). Microplastic in the sediments of a highly eutrophic tropical estuary. *Marine Pollution Bulletin*, *146*(June), 326–335. https://doi.org/10.1016/j.marpolbul.2019.06.042

Andrady, A. L. (2011). Microplastics in the marine environment. *Marine Pollution Bulletin*, *62*(8), 1596–1605. https://doi.org/10.1016/j.marpolbul.2011.05.030

Aslam, H., Ali, T., Mortula, M. M., & Attaelmanan, A. G. (2020). Evaluation of microplastics in beach sediments along the coast of Dubai, UAE. *Marine Pollution Bulletin*, *150*(July 2019), 110739. https://doi.org/10.1016/j.marpolbul.2019.110739

Atwood, E. C., Falcieri, F. M., Piehl, S., Bochow, M., Matthies, M., Franke, J., Carniel, S., Sclavo, M., Laforsch, C., & Siegert, F. (2019). Coastal accumulation of microplastic particles emitted from the Po River, Northern Italy: Comparing remote sensing and hydrodynamic modelling with in situ sample collections. *Marine Pollution Bulletin*, *138*, 561–574. https://doi.org/10.1016/j.marpolbul.2018.11.045

Avio, C. G., Gorbi, S., & Regoli, F. (2017). Plastics and microplastics in the oceans: From emerging pollutants to emerged threat. *Marine Environmental Research*, *128*, 2–11. https://doi.org/10.1016/j.marenvres.2016.05.012

Baldwin, J. R., & Lovvorn, J. R. (1994). Expansion of seagrass habitat by the exotic Zostera japonica, and its use by dabbling ducks and brant in Boundary Bay, British Columbia. *Marine Ecology Progress Series*, *103*(1–2), 119–128. https://doi.org/10.3354/meps103119

Baptista Neto, J. A., de Carvalho, D. G., Medeiros, K., Drabinski, T. L., de Melo, G. V., Silva, R. C. O., Silva, D. C. P., de Sousa Batista, L., Dias, G. T. M., da Fonseca, E. M., & dos Santos Filho, J. R. (2019). The impact of sediment dumping sites on the concentrations of microplastic in the inner continental shelf of Rio de Janeiro/Brazil. *Marine Pollution Bulletin*, *149*(August), 110558. https://doi.org/10.1016/j.marpolbul.2019.110558

Barsanti, M., Garcia-Tenorio, R., Schirone, A., Rozmaric, M., Ruiz-Fernández, A. C., Sanchez-

Cabeza, J. A., Delbono, I., Conte, F., De Oliveira Godoy, J. M., Heijnis, H., Eriksson, M., Hatje, V., Laissaoui, A., Nguyen, H. Q., Okuku, E., Al-Rousan, S. A., Uddin, S., Yii, M. W., & Osvath, I. (2020). Challenges and limitations of the 210Pb sediment dating method: Results from an IAEA modelling interlaboratory comparison exercise. *Quaternary Geochronology*, *59*(April). https://doi.org/10.1016/j.quageo.2020.101093

Bergmann, M., Gutow, L., & Klages, M. (2015). Marine anthropogenic litter. *Marine Anthropogenic Litter*, 1–447. https://doi.org/10.1007/978-3-319-16510-3

Bergmann, M., Wirzberger, V., Krumpen, T., Lorenz, C., Primpke, S., Tekman, M. B., & Gerdts, G. (2017). High Quantities of Microplastic in Arctic Deep-Sea Sediments from the HAUSGARTEN Observatory. *Environmental Science and Technology*, *51*(19), 11000–11010. https://doi.org/10.1021/acs.est.7b03331

Blair, R. M., Waldron, S., & Gauchotte-Lindsay, C. (2019). Average daily flow of microplastics through a tertiary wastewater treatment plant over a ten-month period. *Water Research*, *163*, 114909. https://doi.org/10.1016/j.watres.2019.114909

Blašković, A., Fastelli, P., Čižmek, H., Guerranti, C., & Renzi, M. (2017). Plastic litter in sediments from the Croatian marine protected area of the natural park of TelašČica bay (Adriatic Sea). *Marine Pollution Bulletin*, *114*(1), 583–586. https://doi.org/10.1016/j.marpolbul.2016.09.018

Blumenröder, J., Sechet, P., Kakkonen, J. E., & Hartl, M. G. J. (2017). Microplastic contamination of intertidal sediments of Scapa Flow, Orkney: A first assessment. *Marine Pollution Bulletin*, *124*(1), 112–120. https://doi.org/10.1016/j.marpolbul.2017.07.009

Bonin, D. (2006). *Assessment of Alpine Lakes for Additional Drinking Water Storage in the Capilano, Seymour and Coquitlam Watersheds*.

Bosker, T., Guaita, L., & Behrens, P. (2018). Microplastic pollution on Caribbean beaches in the Lesser Antilles. *Marine Pollution Bulletin*, *133*(May), 442–447. https://doi.org/10.1016/j.marpolbul.2018.05.060

Boucher, J., & Friot, D. (2017). Primary microplastics in the oceans: A global evaluation of sources. In *Primary microplastics in the oceans: A global evaluation of sources*. https://doi.org/10.2305/iucn.ch.2017.01.en

Brandon, J. A., Jones, W., & Ohman, M. D. (2019). Multidecadal increase in plastic particles in coastal ocean sediments. *Science Advances*, *5*(9), 1–7. https://doi.org/10.1126/sciadv.aax0587

Bridson, J. H., Patel, M., Lewis, A., Gaw, S., & Parker, K. (2020). Microplastic contamination in Auckland (New Zealand) beach sediments. *Marine Pollution Bulletin*, *151*(December 2019), 110867. https://doi.org/10.1016/j.marpolbul.2019.110867

Browne, M. A., Crump, P., Niven, S. J., Teuten, E., Tonkin, A., Galloway, T., & Thompson, R. (2011). Accumulation of microplastic on shorelines woldwide: Sources and sinks. *Environmental Science and Technology*, *45*(21), 9175–9179. https://doi.org/10.1021/es201811s

Bucol, L. A., Romano, E. F., Cabcaban, S. M., Siplon, L. M. D., Madrid, G. C., Bucol, A. A., & Polidoro, B. (2020). Microplastics in marine sediments and rabbitfish (Siganus fuscescens) from selected coastal areas of Negros Oriental, Philippines. *Marine Pollution Bulletin*, *150*(October 2019), 110685. https://doi.org/10.1016/j.marpolbul.2019.110685

Carretero, O., Gago, J., & Viñas, L. (2021). From the coast to the shelf: Microplastics in Rías Baixas and Miño River shelf sediments (NW Spain). *Marine Pollution Bulletin*, *162*(December), 111814. https://doi.org/10.1016/j.marpolbul.2020.111814

Chen, M. C., & Chen, T. H. (2020). Spatial and seasonal distribution of microplastics on sandy beaches along the coast of the Hengchun Peninsula, Taiwan. *Marine Pollution Bulletin*, *151*(December 2019), 110861. https://doi.org/10.1016/j.marpolbul.2019.110861

Chen, M., Du, M., Jin, A., Chen, S., Dasgupta, S., Li, J., Xu, H., Ta, K., & Peng, X. (2020). Forty-year pollution history of microplastics in the largest marginal sea of the western Pacific. *Geochemical Perspectives Letters*, 42–47. https://doi.org/10.7185/geochemlet.2012

Chen, Minglong, Jin, M., Tao, P., Wang, Z., Xie, W., Yu, X., & Wang, K. (2018). Assessment of microplastics derived from mariculture in Xiangshan Bay, China. *Environmental Pollution*, *242*, 1146–1156. https://doi.org/10.1016/j.envpol.2018.07.133

Chico-Ortiz, N., Mahu, E., Crane, R., Gordon, C., & Marchant, R. (2020). Microplastics in Ghanaian coastal lagoon sediments: Their occurrence and spatial distribution. *Regional*

Studies in Marine Science, *40*, 101509. https://doi.org/10.1016/j.rsma.2020.101509

Chouchene, K., Prata, J. C., da Costa, J., Duarte, A. C., Rocha-Santos, T., & Ksibi, M. (2021). Microplastics on Barra beach sediments in Aveiro, Portugal. *Marine Pollution Bulletin*, *167*(March). https://doi.org/10.1016/j.marpolbul.2021.112264

Chouchene, K., Rocha-Santos, T., & Ksibi, M. (2020). Types, occurrence, and distribution of microplastics and metals contamination in sediments from south west of Kerkennah archipelago, Tunisia. *Environmental Science and Pollution Research*. https://doi.org/10.1007/s11356-020-09938-z

Cincinelli, A., Scopetani, C., Chelazzi, D., Martellini, T., Pogojeva, M., & Slobodnik, J. (2021). Microplastics in the Black Sea sediments. *Science of the Total Environment*, *760*, 143898. https://doi.org/10.1016/j.scitotenv.2020.143898

Claessens, M., Meester, S. De, Landuyt, L. Van, Clerck, K. De, & Janssen, C. R. (2011a). Occurrence and distribution of microplastics in marine sediments along the Belgian coast. *Marine Pollution Bulletin*, *62*(10), 2199–2204. https://doi.org/10.1016/j.marpolbul.2011.06.030

Claessens, M., Meester, S. De, Landuyt, L. Van, Clerck, K. De, & Janssen, C. R. (2011b). Occurrence and distribution of microplastics in marine sediments along the Belgian coast. *Marine Pollution Bulletin*, *62*(10), 2199–2204. https://doi.org/10.1016/j.marpolbul.2011.06.030

Cluzard, M., Kazmiruk, T. N., Kazmiruk, V. D., & Bendell, L. I. (2015). Intertidal Concentrations of Microplastics and Their Influence on Ammonium Cycling as Related to the Shellfish Industry. *Archives of Environmental Contamination and Toxicology*, *69*(3), 310–319. https://doi.org/10.1007/s00244-015-0156-5

Collicutt, B., Juanes, F., & Dudas, S. E. (2019). Microplastics in juvenile Chinook salmon and their nearshore environments on the east coast of Vancouver Island. *Environmental Pollution*, *244*, 135–142. https://doi.org/10.1016/j.envpol.2018.09.137

Coote, A. (1965). *A Physical and Chemical Study of Tofino Inlet, Vancovuer Island, British Columbia*. University of British Columbia.

Cordova, M., Hadi, T., & Prayudha, B. (2018). Occurrence and abundance of microplastics in coral reef sediment: a case study in Sekotong, Lombok-Indonesia. *Advances in Environmental Sciences*, *10*(1), 23–29. https://doi.org/10.5281/zenodo.1297719

Cordova, M. R., Ulumuddin, Y. I., Purbonegoro, T., & Shiomoto, A. (2021). Characterization of microplastics in mangrove sediment of Muara Angke Wildlife Reserve, Indonesia. *Marine Pollution Bulletin*, *163*(October 2020), 112012. https://doi.org/10.1016/j.marpolbul.2021.112012

Courtene-Jones, W., Quinn, B., Ewins, C., Gary, S. F., & Narayanaswamy, B. E. (2020). Microplastic accumulation in deep-sea sediments from the Rockall Trough. *Marine Pollution Bulletin*, *154*(December 2019), 111092. https://doi.org/10.1016/j.marpolbul.2020.111092

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. *Aquaculture Environment Interactions*, *11*, 357–374. https://doi.org/10.3354/aei00316

Dashtgard, S. E. (2011). Neoichnology of the lower delta plain : Fraser River Delta , British Columbia , Canada : Implications for the ichnology of deltas. *Palaeogeography, Palaeoclimatology, Palaeoecology*, *307*(1–4), 98–108. https://doi.org/10.1016/j.palaeo.2011.05.001

Deng, J., Guo, P., Zhang, X., Su, H., Zhang, Y., Wu, Y., & Li, Y. (2020). Microplastics and accumulated heavy metals in restored mangrove wetland surface sediments at Jinjiang Estuary (Fujian, China). *Marine Pollution Bulletin*, *159*(July), 111482. https://doi.org/10.1016/j.marpolbul.2020.111482

Desforges, J. P. W., Galbraith, M., Dangerfield, N., & Ross, P. S. (2014). Widespread distribution of microplastics in subsurface seawater in the NE Pacific Ocean. *Marine Pollution Bulletin*, *79*(1–2), 94–99. https://doi.org/10.1016/j.marpolbul.2013.12.035

Desforges, J. P. W., Galbraith, M., & Ross, P. S. (2015). Ingestion of Microplastics by Zooplankton in the Northeast Pacific Ocean. *Archives of Environmental Contamination and Toxicology*, *69*(3), 320–330. https://doi.org/10.1007/s00244-015-0172-5

Dodson, G. Z., Shotorban, A. K., Hatcher, P. G., Waggoner, D. C., Ghosal, S., & Noffke, N. (2020). Microplastic fragment and fiber contamination of beach sediments from selected sites in Virginia and North Carolina, USA. *Marine Pollution Bulletin*, *151*(October 2019), 110869. https://doi.org/10.1016/j.marpolbul.2019.110869

Dris, R., Gasperi, J., Saad, M., Mirande, C., & Tassin, B. (2016). Synthetic fibers in atmospheric fallout: A source of microplastics in the environment? In *Marine Pollution Bulletin* (Vol. 104, Issues 1–2, pp. 290–293). https://doi.org/10.1016/j.marpolbul.2016.01.006

Enders, K., Käppler, A., Biniasch, O., Feldens, P., Stollberg, N., Lange, X., Fischer, D., Eichhorn, K. J., Pollehne, F., Oberbeckmann, S., & Labrenz, M. (2019). Tracing microplastics in aquatic environments based on sediment analogies. *Scientific Reports*, *9*(1), 1–15. https://doi.org/10.1038/s41598-019-50508-2

Feng, S., Lu, H., Tian, P., Xue, Y., Lu, J., Tang, M., & Feng, W. (2020). Analysis of microplastics in a remote region of the Tibetan Plateau: Implications for natural environmental response to human activities. *Science of the Total Environment*, *739*, 140087. https://doi.org/10.1016/j.scitotenv.2020.140087

Filgueiras, A. V., Gago, J., Campillo, J. A., & León, V. M. (2019). Microplastic distribution in surface sediments along the Spanish Mediterranean continental shelf. *Environmental Science and Pollution Research*, *26*(21), 21264–21273. https://doi.org/10.1007/s11356-019-05341-5

Firdaus, M., Trihadiningrum, Y., & Lestari, P. (2020). Microplastic pollution in the sediment of Jagir Estuary, Surabaya City, Indonesia. *Marine Pollution Bulletin*, *150*(September 2019), 110790. https://doi.org/10.1016/j.marpolbul.2019.110790

Flaherty, M., Reid, G., Chopin, T., & Latham, E. (2019). Public attitudes towards marine aquaculture in Canada: insights from the Pacific and Atlantic coasts. *Aquaculture International*, *27*(1), 9–32. https://doi.org/10.1007/s10499-018-0312-9

Fok, L., & Cheung, P. K. (2015). Hong Kong at the Pearl River Estuary: A hotspot of microplastic pollution. *Marine Pollution Bulletin*, *99*(1–2), 112–118. https://doi.org/10.1016/j.marpolbul.2015.07.050

Franzellitti, S., Canesi, L., Auguste, M., Wathsala, R. H. G. R., & Fabbri, E. (2019). Microplastic

exposure and effects in aquatic organisms: A physiological perspective. *Environmental Toxicology and Pharmacology*, *68*(March), 37–51. https://doi.org/10.1016/j.etap.2019.03.009

Fraser, M. A., Chen, L., Ashar, M., Huang, W., Zeng, J., Zhang, C., & Zhang, D. (2020). Occurrence and distribution of microplastics and polychlorinated biphenyls in sediments from the Qiantang River and Hangzhou Bay, China. *Ecotoxicology and Environmental Safety*, *196*(December 2019), 110536. https://doi.org/10.1016/j.ecoenv.2020.110536

Free, C. M., Jensen, O. P., Mason, S. A., Eriksen, M., Williamson, N. J., & Boldgiv, B. (2014). High-levels of microplastic pollution in a large, remote, mountain lake. *Marine Pollution Bulletin*, *85*(1), 156–163. https://doi.org/10.1016/j.marpolbul.2014.06.001

Freeland, H. J., Crawford, W. R., & Thomson, R. E. (1984). Currents along the pacific coast of canada. *Atmosphere - Ocean*, *22*(2), 151–172. https://doi.org/10.1080/07055900.1984.9649191

Frias, J. P. G. L., Gago, J., Otero, V., & Sobral, P. (2016). Microplastics in coastal sediments from Southern Portuguese shelf waters. *Marine Environmental Research*, *114*(2016), 24–30. https://doi.org/10.1016/j.marenvres.2015.12.006

Frouin, H., Dangerfield, N., Macdonald, R. W., Galbraith, M., Crewe, N., Shaw, P., Mackas, D., & Ross, P. S. (2013). Partitioning and bioaccumulation of pcbs and pbdes in marine plankton from the strait of Georgia, British Columbia, Canada. *Progress in Oceanography*, *115*, 65–75. https://doi.org/10.1016/j.pocean.2013.05.023

Gago, J., Carretero, O., Filgueiras, A. V, & Viñas, L. (2017). *Synthetic microfibers in the marine environment: A review on their occurrence in seawater and sediments*. https://doi.org/10.1016/j.marpolbul.2017.11.070

Geyer, R., Jambeck, J., & Law, K. (2017). Production, Use, And Fate Of All Plastics Ever Made. *Science Advances*, *3*(7), 25–29.

Gies, E. A., LeNoble, J. L., Noël, M., Etemadifar, A., Bishay, F., Hall, E. R., & Ross, P. S. (2018a). Retention of microplastics in a major secondary wastewater treatment plant in Vancouver, Canada. *Marine Pollution Bulletin*, *133*(March), 553–561. https://doi.org/10.1016/j.marpolbul.2018.06.006

Gies, E. A., LeNoble, J. L., Noël, M., Etemadifar, A., Bishay, F., Hall, E. R., & Ross, P. S. (2018b). Retention of microplastics in a major secondary wastewater treatment plant in Vancouver, Canada. *Marine Pollution Bulletin*, *133*(June), 553–561. https://doi.org/10.1016/j.marpolbul.2018.06.006

Graca, B., Szewc, K., Zakrzewska, D., Dołęga, A., & Szczerbowska-Boruchowska, M. (2017). Sources and fate of microplastics in marine and beach sediments of the Southern Baltic Sea—a preliminary study. *Environmental Science and Pollution Research*, *24*(8), 7650–7661. https://doi.org/10.1007/s11356-017-8419-5

Gray, A. D., Wertz, H., Leads, R. R., & Weinstein, J. E. (2018). Microplastic in two South Carolina Estuaries: Occurrence, distribution, and composition. *Marine Pollution Bulletin*, *128*(February), 223–233. https://doi.org/10.1016/j.marpolbul.2018.01.030

Guerranti, C., Cannas, S., Scopetani, C., Fastelli, P., Cincinelli, A., & Renzi, M. (2017). Plastic litter in aquatic environments of Maremma Regional Park (Tyrrhenian Sea, Italy): Contribution by the Ombrone river and levels in marine sediments. *Marine Pollution Bulletin*, *117*(1–2), 366–370. https://doi.org/10.1016/j.marpolbul.2017.02.021

Haave, M., Lorenz, C., Primpke, S., & Gerdts, G. (2019a). Different stories told by small and large microplastics in sediment - first report of microplastic concentrations in an urban recipient in Norway. *Marine Pollution Bulletin*, *141*(February), 501–513. https://doi.org/10.1016/j.marpolbul.2019.02.015

Haave, M., Lorenz, C., Primpke, S., & Gerdts, G. (2019b). Different stories told by small and large microplastics in sediment - first report of microplastic concentrations in an urban recipient in Norway. *Marine Pollution Bulletin*, *141*(November 2018), 501–513. https://doi.org/10.1016/j.marpolbul.2019.02.015

Hahladakis, J. N., Velis, C. A., Weber, R., Iacovidou, E., & Purnell, P. (2018). An overview of chemical additives present in plastics: Migration, release, fate and environmental impact during their use, disposal and recycling. In *Journal of Hazardous Materials* (Vol. 344, pp. 179–199). Elsevier B.V. https://doi.org/10.1016/j.jhazmat.2017.10.014

Hall, N. M., Berry, · K L E, Rintoul, · L, & Hoogenboom, · M O. (2015). Microplastic ingestion by scleractinian corals. *Mar Biol*, *3*, 725–732. https://doi.org/10.1007/s00227-015-2619-7

Hanvey, J. S., Lewis, P. J., Lavers, J. L., Crosbie, N. D., Pozo De, K., & Clarke, B. O. (2017). *A review of analytical techniques for quantifying microplastics in sediments*. https://doi.org/10.1039/c6ay02707e

Harris, P. T. (2020). The fate of microplastic in marine sedimentary environments: A review and synthesis. *Marine Pollution Bulletin*, *158*(June), 111398. https://doi.org/10.1016/j.marpolbul.2020.111398

Hidalgo-Ruz, V., Gutow, L., Thompson, R. C., & Thiel, M. (2013). Microplastics in the marine environment: A review of the methods used for identification and quantification. *… Science & Technology*, *46*, 3060–3075. https://doi.org/10.1021/es2031505

Horton, A. A., Svendsen, C., Williams, R. J., Spurgeon, D. J., & Lahive, E. (2017). Large microplastic particles in sediments of tributaries of the River Thames, UK – Abundance, sources and methods for effective quantification. *Marine Pollution Bulletin*, *114*(1), 218–226. https://doi.org/10.1016/j.marpolbul.2016.09.004

Irabien, M. J., Cearreta, A., Gómez-Arozamena, J., & García-Artola, A. (2020). Holocene vs Anthropocene sedimentary records in a human-altered estuary: The Pasaia case (northern Spain). *Marine Geology*, *429*(July), 106292. https://doi.org/10.1016/j.margeo.2020.106292

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*(6223), 768–771. https://doi.org/10.1126/science.1260352

Johannessen, D. I., & Ross, P. S. (2002). Late-run sockeye at risk : An overview of environmental contaminants in Fraser River salmon habitat. *Can. Tech. Rep. Fish. Aquat. Sci.*, *2429*(February), x + 108.

Johannessen, S., Macdonald, R., Burd, B., van Roodselaar, A., & Bertold, S. (2015). Local environmental conditions determine the footprint of municipal effluent in coastal waters: A case study in the Strait of Georgia, British Columbia. *Science of the Total Environment*, *508*, 228–239. https://doi.org/10.1016/j.scitotenv.2014.11.096

Kanhai, L. D. K., Johansson, C., Frias, J. P. G. L., Gardfeldt, K., Thompson, R. C., & O'Connor, I. (2019). Deep sea sediments of the Arctic Central Basin: A potential sink for microplastics. *Deep-Sea Research Part I: Oceanographic Research Papers*, *145*(June 2018), 137–142.

https://doi.org/10.1016/j.dsr.2019.03.003

Kazmiruk, T. N., Kazmiruk, V. D., & Bendell, L. I. (2018). Abundance and distribution of microplastics within surface sediments of a key shellfish growing region of Canada. *PLoS ONE*, *13*(5), 1–16. https://doi.org/10.1371/journal.pone.0196005

Kirchner, G. (2011). 210Pb as a tool for establishing sediment chronologies: Examples of potentials and limitations of conventional dating models. In *Journal of Environmental Radioactivity* (Vol. 102, Issue 5, pp. 490–494). https://doi.org/10.1016/j.jenvrad.2010.11.010

Kor, K., Ghazilou, A., & Ershadifar, H. (2020). Microplastic pollution in the littoral sediments of the northern part of the Oman Sea. *Marine Pollution Bulletin*, *155*(April), 111166. https://doi.org/10.1016/j.marpolbul.2020.111166

Krüger, L., Casado-Coy, N., Valle, C., Ramos, M., Sánchez-Jerez, P., Gago, J., Carretero, O., Beltran-Sanahuja, A., & Sanz-Lazaro, C. (2020). Plastic debris accumulation in the seabed derived from coastal fish farming. *Environmental Pollution*, *257*. https://doi.org/10.1016/j.envpol.2019.113336

Laglbauer, B. J. L., Franco-Santos, R. M., Andreu-Cazenave, M., Brunelli, L., Papadatou, M., Palatinus, A., Grego, M., & Deprez, T. (2014). Macrodebris and microplastics from beaches in Slovenia. *Marine Pollution Bulletin*, *89*(1–2), 356–366. https://doi.org/10.1016/j.marpolbul.2014.09.036

LeBlond, P. H. (1983). The Strait of Georgia: functional anatomy of a coastal sea. In *Canadian Journal of Fisheries and Aquatic Sciences* (Vol. 40, Issue 7, pp. 1033–1063). https://doi.org/10.1139/f83-128

Lebreton, L., & Andrady, A. (2019). Future scenarios of global plastic waste generation and disposal. *Palgrave Communications*, *5*(1), 1–11. https://doi.org/10.1057/s41599-018-0212-7

Lechthaler, S., Schwarzbauer, J., Reicherter, K., Stauch, G., & Schüttrumpf, H. (2020). Regional study of microplastics in surface waters and deep sea sediments south of the Algarve Coast. *Regional Studies in Marine Science*, *40*, 101488. https://doi.org/10.1016/j.rsma.2020.101488

Lewis, S. L., & Maslin, M. A. (2015). Defining the Anthropocene. *Nature*, *519*(7542), 171–180. https://doi.org/10.1038/nature14258

Li, J., Huang, W., Xu, Y., Jin, A., Zhang, D., & Zhang, C. (2020). Microplastics in sediment cores as indicators of temporal trends in microplastic pollution in Andong salt marsh, Hangzhou Bay, China. *Regional Studies in Marine Science*, *35*, 101149. https://doi.org/10.1016/j.rsma.2020.101149

Li, Yizheng, Zhang, Y., Chen, G., Xu, K., Gong, H., Huang, K., Yan, M., & Wang, J. (2021). Microplastics in surface waters and sediments from Guangdong coastal areas, South China. *Sustainability (Switzerland)*, *13*(5), 1–15. https://doi.org/10.3390/su13052691

Li, Yubo, Lu, Z., Zheng, H., Wang, J., & Chen, C. (2020). Microplastics in surface water and sediments of Chongming Island in the Yangtze Estuary, China. *Environmental Sciences Europe*, *32*(1). https://doi.org/10.1186/s12302-020-0297-7

Liebezeit, G., & Dubaish, F. (2012). Microplastics in beaches of the East Frisian Islands Spiekeroog and Kachelotplate. *Bulletin of Environmental Contamination and Toxicology*, *89*(1), 213–217. https://doi.org/10.1007/s00128-012-0642-7

Lloret, J., Pedrosa-Pamies, R., Vandal, N., Rorty, R., Ritchie, M., McGuire, C., Chenoweth, K., & Valiela, I. (2021). Salt marsh sediments act as sinks for microplastics and reveal effects of current and historical land use changes. *Environmental Advances*, *4*(April), 100060. https://doi.org/10.1016/j.envadv.2021.100060

Loughlin, C., Marques, A. R., Morrison, L., & Morley, A. (2021). The role of oceanographic processes and sedimentological settings on the deposition of microplastics in marine sediment : Icelandic waters. *Marine Pollution Bulletin*, *164*(December 2020), 111976. https://doi.org/10.1016/j.marpolbul.2021.111976

Luo, H., Xiang, Y., He, D., Li, Y., Zhao, Y., Wang, S., & Pan, X. (2019). *Leaching behavior of fluorescent additives from microplastics and the toxicity of leachate to Chlorella vulgaris*. https://doi.org/10.1016/j.scitotenv.2019.04.401

Lusher, A., Hollman, P., & Mendozal, J. (2017). Microplastics in fisheries and aquaculture: status of knowledge on their occurrence and implications for aquatic organisms and food safety. In *FAO Fisheries and Aquaculture Technical Paper 615*. https://doi.org/978-92-5-

109882-0

Macdonald, R. W., Macdonald, D. M., O'Brien, M. C., & Gobeil, C. (1991). Accumulation of heavy metals (Pb, Zn, Cu, Cd), carbon and nitrogen in sediments from Strait of Georgia, B.C., Canada. *Marine Chemistry*, *34*(1–2), 109–135. https://doi.org/10.1016/0304-4203(91)90017-Q

Maes, T., Van der Meulen, M. D., Devriese, L. I., Leslie, H. A., Huvet, A., Frère, L., Robbens, J., & Vethaak, A. D. (2017). Microplastics baseline surveys at the water surface and in sediments of the North-East Atlantic. *Frontiers in Marine Science*, *4*(MAY), 1–14. https://doi.org/10.3389/fmars.2017.00135

Marine Strategy Framework Directive. 2008. Annex III. 2017. Accessed in 2021.

Martin, J., Lusher, A., Thompson, R. C., & Morley, A. (2017). The Deposition and Accumulation of Microplastics in Marine Sediments and Bottom Water from the Irish Continental Shelf. *Scientific Reports*, *7*(1), 1–9. https://doi.org/10.1038/s41598-017-11079-2

Masiá, P., Ardura, A., & Garcia-Vazquez, E. (2019). Microplastics in special protected areas for migratory birds in the Bay of Biscay. *Marine Pollution Bulletin*, *146*(July), 993–1001. https://doi.org/10.1016/j.marpolbul.2019.07.065

Mathalon, A., & Hill, P. (2014a). Microplastic fibers in the intertidal ecosystem surrounding Halifax Harbor, Nova Scotia. *Marine Pollution Bulletin*, *81*(1), 69–79. https://doi.org/10.1016/j.marpolbul.2014.02.018

Mathalon, A., & Hill, P. (2014b). Microplastic fibers in the intertidal ecosystem surrounding Halifax Harbor, Nova Scotia. *Marine Pollution Bulletin*, *81*(1), 69–79. https://doi.org/10.1016/j.marpolbul.2014.02.018

Matsuguma, Y., Takada, H., Kumata, H., Kanke, H., Sakurai, S., Suzuki, T., Itoh, M., Okazaki, Y., Boonyatumanond, R., Zakaria, M. P., Weerts, S., & Newman, B. (2017). Microplastics in Sediment Cores from Asia and Africa as Indicators of Temporal Trends in Plastic Pollution. *Archives of Environmental Contamination and Toxicology*, *73*(2), 230–239. https://doi.org/10.1007/s00244-017-0414-9

Mayoma, B. S., Sørensen, C., Shashoua, Y., & Khan, F. R. (2020). Microplastics in beach

sediments and cockles (Anadara antiquata) along the Tanzanian coastline. *Bulletin of Environmental Contamination and Toxicology*, *105*(4), 513–521. https://doi.org/10.1007/s00128-020-02991-x

McEachern, K., Alegria, H., Kalagher, A. L., Hansen, C., Morrison, S., & Hastings, D. (2019). Microplastics in Tampa Bay, Florida: Abundance and variability in estuarine waters and sediments. *Marine Pollution Bulletin*, *148*(July), 97–106. https://doi.org/10.1016/j.marpolbul.2019.07.068

Mistri, M., Infantini, V., Scoponi, M., Granata, T., Moruzzi, L., Massara, F., De Donati, M., & Munari, C. (2017). Small plastic debris in sediments from the Central Adriatic Sea: Types, occurrence and distribution. *Marine Pollution Bulletin*, *124*(1), 435–440. https://doi.org/10.1016/j.marpolbul.2017.07.063

Mistri, M., Scoponi, M., Granata, T., Moruzzi, L., Massara, F., & Munari, C. (2020). Types, occurrence and distribution of microplastics in sediments from the northern Tyrrhenian Sea. *Marine Pollution Bulletin*, *153*(January), 111016. https://doi.org/10.1016/j.marpolbul.2020.111016

Mohamed Nor, N. H., & Obbard, J. P. (2014). Microplastics in Singapore's coastal mangrove ecosystems. *Marine Pollution Bulletin*, *79*(1–2), 278–283. https://doi.org/10.1016/j.marpolbul.2013.11.025

Mu, J., Qu, L., Jin, F., Zhang, S., Fang, C., Ma, X., Zhang, W., Huo, C., Cong, Y., & Wang, J. (2019). Abundance and distribution of microplastics in the surface sediments from the northern Bering and Chukchi Seas. *Environmental Pollution*, *245*, 122–130. https://doi.org/10.1016/j.envpol.2018.10.097

Nel, H. A., & Froneman, P. W. (2015). A quantitative analysis of microplastic pollution along the south-eastern coastline of South Africa. *Marine Pollution Bulletin*, *101*(1), 274–279. https://doi.org/10.1016/j.marpolbul.2015.09.043

Ngo, P. L., Pramanik, B. K., Shah, K., & Roychand, R. (2019). Pathway, classification and removal efficiency of microplastics in wastewater treatment plants. *Environmental Pollution*, *255*. https://doi.org/10.1016/j.envpol.2019.113326

Norman, E. S. (2013). Who's counting? Spatial politics, ecocolonisation and the politics of

calculation in Boundary Bay. In *Area* (Vol. 45, Issue 2, pp. 179–187). https://doi.org/10.1111/area.12000

Okey, T. A., & Dallimore, A. (2015). *Overview of the Climate and Oceanography of the West Coast of Vancouver Island, Canada*. *March 2011*.

Pagter, E., Frias, J., Kavanagh, F., & Nash, R. (2020). Varying levels of microplastics in benthic sediments within a shallow coastal embayment. *Estuarine, Coastal and Shelf Science*, *243*(February). https://doi.org/10.1016/j.ecss.2020.106915

Peng, G., Zhu, B., Yang, D., Su, L., Shi, H., & Li, D. (2017). Microplastics in sediments of the Changjiang Estuary, China. *Environmental Pollution*, *225*, 283–290. https://doi.org/10.1016/j.envpol.2016.12.064

Peng, J., Wang, J., & Cai, L. (2017). Current understanding of microplastics in the environment: Occurrence, fate, risks, and what we should do. In *Integrated Environmental Assessment and Management* (Vol. 13, Issue 3, pp. 476–482). https://doi.org/10.1002/ieam.1912

Peng, X., Chen, M., Chen, S., Dasgupta, S., Xu, H., Ta, K., Du, M., Li, J., Guo, Z., & Bai, S. (2018). Microplastics contaminate the deepest part of the world's ocean. *Geochemical Perspectives Letters*, *9*, 1–5. https://doi.org/10.7185/geochemlet.1829

PlasticsEurope, Plastics—The Facts 2012: An Analysis of European Plastics Production, Demand and Waste Data for 2011 (PlasticsEurope, 2012); http://www.plasticseurope.org/

Postlethwaite, V. R., McGowan, A. E., Kohfeld, K. E., Robinson, C. L. K., & Pellatt, M. G. (2018). Low blue carbon storage in eelgrass (Zostera marina) meadows on the Pacific Coast of Canada. *PLoS ONE*, *13*(6), 1–18. https://doi.org/10.1371/journal.pone.0198348

Province of British Columbia. Aquaculture. Accessed in 2021. https://www2.gov.bc.ca/gov/content/industry/agriculture-seafood/fisheries-and-aquaculture/aquaculture

Qiging, C., Allgeier, A., Yin, D., & Hollert, H. (2019). *Leaching of endocrine disrupting chemicals from marine microplastics and mesoplastics under common life stress conditions*. https://doi.org/10.1016/j.envint.2019.104938

Reed, S., Clark, M., Thompson, R., & Hughes, K. A. (2018). Microplastics in marine sediments

near Rothera Research Station, Antarctica. *Marine Pollution Bulletin*, *133*(December 2017), 460–463. https://doi.org/10.1016/j.marpolbul.2018.05.068

Ronda, A. C., Arias, A. H., Oliva, A. L., & Marcovecchio, J. E. (2019). Synthetic microfibers in marine sediments and surface seawater from the Argentinean continental shelf and a Marine Protected Area. *Marine Pollution Bulletin*, *149*(June), 110618. https://doi.org/10.1016/j.marpolbul.2019.110618

Sandre, F., Dromard, C. R., Menach, K. Le, Bouchon-Navaro, Y., Cordonnie, S., Tapie, N., Budzinski, H., & Bouchon, C. (2019). MICROPLASTIC DISTRIBUTION and DETECTION of CHLORDECONE on MICROPLASTICS in MARINE SEDIMENTS in GUADELOUPE: A PRELIMINARY STUDY. *Gulf and Caribbean Research*, *30*(1), GCFI8–GCFI14. https://doi.org/10.18785/GCR.3001.14

Schröder, K., Kossel, E., & Lenz, M. (2021). Microplastic abundance in beach sediments of the Kiel Fjord, Western Baltic Sea. *Environmental Science and Pollution Research*, *28*(21), 26515–26528. https://doi.org/10.1007/s11356-020-12220-x

Shabaka, S. H., Ghobashy, M., & Marey, R. S. (2019). Identification of marine microplastics in Eastern Harbor, Mediterranean Coast of Egypt, using differential scanning calorimetry. *Marine Pollution Bulletin*, *142*(January), 494–503. https://doi.org/10.1016/j.marpolbul.2019.03.062

Shepperd, J. (1981). *Development of a Salt Marsh on the Fraser Delta at Boundary Bay, British Columbia, Canada*. The University of British Columbia.

Sighicelli, M., Pietrelli, L., Lecce, F., Iannilli, V., Falconieri, M., Coscia, L., Di Vito, S., Nuglio, S., & Zampetti, G. (2018). Microplastic pollution in the surface waters of Italian Subalpine Lakes. *Environmental Pollution*, *236*, 645–651. https://doi.org/10.1016/j.envpol.2018.02.008

Statistics Canada. 2016 Census Data. 2016. Accessed in 2021.

Stolte, A., Forster, S., Gerdts, G., & Schubert, H. (2015). Microplastic concentrations in beach sediments along the German Baltic coast. *Marine Pollution Bulletin*, *99*(1–2), 216–229. https://doi.org/10.1016/j.marpolbul.2015.07.022

Sun, J., Dai, X., Wang, Q., van Loosdrecht, M. C. M., & Ni, B. J. (2019). Microplastics in wastewater treatment plants: Detection, occurrence and removal. *Water Research*, *152*, 21–37. https://doi.org/10.1016/j.watres.2018.12.050

Swinbanks, D., & Murray, J. (1981). Biosedimentological zonation of Boundary Bay tidal flatd, Fraser River Delta, British Columbia. *Ocean Research*, 201–237.

Talvitie, J., Mikola, A., Koistinen, A., & Setälä, O. (2017). Solutions to microplastic pollution – Removal of microplastics from wastewater effluent with advanced wastewater treatment technologies. *Water Research*, *123*, 401–407. https://doi.org/10.1016/j.watres.2017.07.005

Taylor, M. L., Gwinnett, C., Robinson, L. F., & Woodall, L. C. (2016). Plastic microfibre ingestion by deep-sea organisms. *Scientific Reports*, *6*(May), 1–9. https://doi.org/10.1038/srep33997

Tsang, Y. Y., Mak, C. W., Liebich, C., Lam, S. W., Sze, E. T. P., & Chan, K. M. (2017). Microplastic pollution in the marine waters and sediments of Hong Kong. *Marine Pollution Bulletin*, *115*(1–2), 20–28. https://doi.org/10.1016/j.marpolbul.2016.11.003

Turner, L. J., & Delorme, L. D. (1996). Assessment of 210Pb data from Canadian lakes using the CIC and CRS models. *Environmental Geology*, *28*(2), 78–87. https://doi.org/10.1007/s002540050080

Uddin, S., Fowler, S. W., & Saeed, T. (2020). Microplastic particles in the Persian/Arabian Gulf – A review on sampling and identification. *Marine Pollution Bulletin*, *154*(February), 111100. https://doi.org/10.1016/j.marpolbul.2020.111100

Uddin, S., Fowler, S. W., Uddin, M. F., Behbehani, M., & Naji, A. (2021). A review of microplastic distribution in sediment profiles. *Marine Pollution Bulletin*, *163*(December 2020), 111973. https://doi.org/10.1016/j.marpolbul.2021.111973

Van Cauwenberghe, L., Devriese, L., Galgani, F., Robbens, J., & Janssen, C. R. (2015). Microplastics in sediments: A review of techniques, occurrence and effects. *Marine Environmental Research*, *111*, 5–17. https://doi.org/10.1016/j.marenvres.2015.06.007

Vaughan, R., Turner, S. D., & Rose, N. L. (2017). Microplastics in the sediments of a UK urban lake. *Environmental Pollution*, *229*, 10–18. https://doi.org/10.1016/j.envpol.2017.05.057

Veerasingam, S., Ranjani, M., Venkatachalapathy, R., Bagaev, A., Mukhanov, V., Litvinyuk, D.,

Mugilarasan, M., Gurumoorthi, K., Guganathan, L., Aboobacker, V. M., & Vethamony, P. (2020). Contributions of Fourier transform infrared spectroscopy in microplastic pollution research: A review. *Critical Reviews in Environmental Science and Technology*, *0*(0), 1–63. https://doi.org/10.1080/10643389.2020.1807450

Velasco, A. de J. N., Rard, L., Blois, W., Lebrun, D., Lebrun, F., Pothe, F., & Stoll, S. (2020). Microplastic and fibre contamination in a remote Mountain lake in Switzerland. *Water (Switzerland)*, *12*(9), 1–16. https://doi.org/10.3390/W12092410

Vianello, A., Boldrin, A., Guerriero, P., Moschino, V., Rella, R., Sturaro, A., & Da Ros, L. (2013). Microplastic particles in sediments of Lagoon of Venice, Italy: First observations on occurrence, spatial patterns and identification. *Estuarine, Coastal and Shelf Science*, *130*, 54–61. https://doi.org/10.1016/j.ecss.2013.03.022

Waldichuk, M. (1957). *Physical Oceanography of the Strait of Georgia , British Colurnbia*. *205*.

Waldichuk, M. (1983). Pollution in the strait of Georgia: A review. *Canadian Journal of Fisheries and Aquatic Sciences*, *40*(7), 1142–1167. https://doi.org/10.1139/f83-132

Wang, Y., Zou, X., Peng, C., Qiao, S., Wang, T., Yu, W., Khokiattiwong, S., & Kornkanitnan, N. (2020). Occurrence and distribution of microplastics in surface sediments from the Gulf of Thailand. *Marine Pollution Bulletin*, *152*(January), 110916. https://doi.org/10.1016/j.marpolbul.2020.110916

Welden, N. A., & Cowie, P. R. (2017). Degradation of common polymer ropes in a sublittoral marine environment. *Marine Pollution Bulletin*, *118*(1–2), 248–253. https://doi.org/10.1016/j.marpolbul.2017.02.072

Wessel, C. C., Lockridge, G. R., Battiste, D., & Cebrian, J. (2016). Abundance and characteristics of microplastics in beach sediments: Insights into microplastic accumulation in northern Gulf of Mexico estuaries. *Marine Pollution Bulletin*, *109*(1), 178–183. https://doi.org/10.1016/j.marpolbul.2016.06.002

Willis, K. A., Eriksen, R., Wilcox, C., & Hardesty, B. D. (2017). Microplastic distribution at different sediment depths in an urban estuary. *Frontiers in Marine Science*, *4*(DEC), 1–8. https://doi.org/10.3389/fmars.2017.00419

Woodall, L., Sanchez-Vidal, A., Canals, M., Paterson, G., Coppock, R., Sleight, V., Calafat, A., Rogers, A., Narayanaswamy, B., & Thompson, R. (2014). *The deep sea is a major sink for microplastic debris*.

Wright, S. L., Thompson, R. C., & Galloway, T. S. (2013). The physical impacts of microplastics on marine organisms: A review. *Environmental Pollution*, *178*, 483–492. https://doi.org/10.1016/j.envpol.2013.02.031

Wu, F., Wang, Y., Leung, J. Y. S., Huang, W., Zeng, J., Tang, Y., Chen, J., Shi, A., Yu, X., Xu, X., Zhang, H., & Cao, L. (2020). Accumulation of microplastics in typical commercial aquatic species: A case study at a productive aquaculture site in China. *Science of the Total Environment*, *708*(36), 135432. https://doi.org/10.1016/j.scitotenv.2019.135432

Xu, J. L., Thomas, K. V., Luo, Z., & Gowen, A. A. (2019). FTIR and Raman imaging for microplastics analysis: State of the art, challenges and prospects. *TrAC - Trends in Analytical Chemistry*, *119*, 115629. https://doi.org/10.1016/j.trac.2019.115629

Yu, Q., Hu, X., Yang, B., Zhang, G., Wang, J., & Ling, W. (2020). Distribution, abundance and risks of microplastics in the environment. *Chemosphere*, *249*, 126059. https://doi.org/10.1016/j.chemosphere.2020.126059

Yunker, M. B., Macdonald, R. W., Goyette, D., Paton, D. W., Fowler, B. R., Sullivan, D., & Boyd, J. (1999). Natural and anthropogenic inputs of hydrocarbons to the Strait of Georgia. *Science of the Total Environment*, *225*(3), 181–209. https://doi.org/10.1016/S0048-9697(98)00362-3

Zhang, C., Zhou, H., Cui, Y., Wang, C., Li, Y., & Zhang, D. (2019). Microplastics in offshore sediment in the Yellow Sea and East China Sea, China. *Environmental Pollution*, *244*(October), 827–833. https://doi.org/10.1016/j.envpol.2018.10.102

Zhang, D., Liu, X., Huang, W., Li, J., Wang, C., Zhang, D., & Zhang, C. (2020). Microplastic pollution in deep-sea sediments and organisms of the Western Pacific Ocean. *Environmental Pollution*, *259*, 113948. https://doi.org/10.1016/j.envpol.2020.113948

Zhang, H. (2017). Transport of microplastics in coastal seas. *Estuarine, Coastal and Shelf Science*, *199*, 74–86. https://doi.org/10.1016/j.ecss.2017.09.032

Zhang, S., Wang, J., Liu, X., Qu, F., Wang, X., Wang, X., Li, Y., & Sun, Y. (2019). Microplastics in the environment: A review of analytical methods, distribution, and biological effects. *TrAC - Trends in Analytical Chemistry*, *111*, 62–72. https://doi.org/10.1016/j.trac.2018.12.002

Zhang, Y., Kang, S., Allen, S., Allen, D., Gao, T., & Sillanpää, M. (2020). Atmospheric microplastics: A review on current status and perspectives. *Earth-Science Reviews*, *203*(December 2019), 103118. https://doi.org/10.1016/j.earscirev.2020.103118

Zhao, J., Ran, W., Teng, J., Liu, Y., Liu, H., Yin, X., Cao, R., & Wang, Q. (2018). Microplastic pollution in sediments from the Bohai Sea and the Yellow Sea, China. *Science of the Total Environment*, *640–641*, 637–645. https://doi.org/10.1016/j.scitotenv.2018.05.346

Zheng, Y., Li, J., Cao, W., Jiang, F., Zhao, C., Ding, H., Wang, M., Gao, F., & Sun, C. (2020). Vertical distribution of microplastics in bay sediment reflecting effects of sedimentation dynamics and anthropogenic activities. *Marine Pollution Bulletin*, *152*(January), 110885. https://doi.org/10.1016/j.marpolbul.2020.110885

Ziajahromi, S., Neale, P. A., & Leusch, F. D. L. (2016). Wastewater treatment plant effluent as a source of microplastics: review of the fate, chemical interactions and potential risks to aquatic organisms. *Water Science and Technology*, *74*(10), 2253–2269. https://doi.org/10.2166/wst.2016.414

Ziajahromi, S., Neale, P. A., Rintoul, L., & Leusch, F. D. L. (2017a). Wastewater treatment plants as a pathway for microplastics: Development of a new approach to sample wastewater-based microplastics. *Water Research*, *112*, 93–99. https://doi.org/10.1016/j.watres.2017.01.042

Ziajahromi, S., Neale, P. A., Rintoul, L., & Leusch, F. D. L. (2017b). Wastewater treatment plants as a pathway for microplastics: Development of a new approach to sample wastewater-based microplastics. *Water Research*, *112*, 93–99. https://doi.org/10.1016/j.watres.2017.01.042

Zobkov, M., & Esiukova, E. (2017). Microplastics in Baltic bottom sediments: Quantification procedures and first results. *Marine Pollution Bulletin*, *114*(2), 724–732. https://doi.org/10.1016/j.marpolbul.2016.10.060

Appendix A. Visual Identification of MPs based on Hidalgo-Ruiz et al. (2012)

Visual examination is an obligatory step in MP studies. Careful visual sorting of residues is necessary to separate the plastic particles from other materials, such as organic debris (shell fragments, animal parts, dried algae, or seagrasses, etc.) and other items. This is done by direct examination of the sample by the naked eye or with the aid of a dissecting microscope. Samples can be preserved in their original form without initial sorting, or they can be immediately sorted to store only the plastics from the original sample. Plastics separated from the sample should be dried and kept in a dark and temperature-controlled environment (stable room temperature) to reduce degradation during storage. To avoid misidentification and underestimation of microplastics it is necessary to standardize the plastic particle selection, following certain criteria to guarantee proper identification. This is particularly important when it is not possible to use more accurate methods, such as Fourier transform infrared spectroscopy (FT-IR). Pieces of microplastics toward the larger end of the size range (>1 mm) can to some extent be visually distinguished according to the following criteria: no cellular or organic structures are visible, fibers should be equally thick throughout their entire length, particles must present clear and homogeneous colors, and if they are transparent or white, they must be examined under high magnification and a fluorescence microscope.

Appendix B. Microplastic Sizes.

Grice Bay - GB REF

Midpoint Depth (cm)	Fiber Colour	Microfiber Size (mm)
		Length (mm)
0.5	Blue	2.17
0.5	Blue	2.39
0.5	Black	3.06
0.5	Black	0.70
0.5	Black	1.47
0.5	Black	1.30
0.5	Black	0.81
0.5	Black	3.46
0.5	Black	4.15
0.5	Grey	2.05
0.5	Grey	1.04
3.5	Green	0.60
3.5	Black	3.13
3.5	Black	2.05
7.5	Black	0.58
7.5	Black	2.22
7.5	Black	0.68
7.5	Black	0.24
7.5	Black	4.02
7.5	Black	2.85
11.5	Black	1.66
11.5	Black	0.95
11.5	Black	1.13
11.5	Black	2.62
11.5	Black	2.11
15.5	Black	1.14
15.5	Black	3.08
15.5	Black	1.10
15.5	Black	0.58
15.5	Grey	2.06
15.5	Grey	1.10
19.5	Black	1.29

23.5	Black	1.95
23.5	Red	1.07
23.5	Blue	2.94
27.5	Black	0.43

Kennedy Cove – KC3

		Microfiber Size (mm)
Midpoint Depth (cm)	**Fiber Colour**	**Length (mm)**
0.5	Blue	2.48
0.5	Blue	1.46
0.5	Black	0.94
0.5	Black	1.59
0.5	Black	2.63
0.5	Black	1.49
0.5	Green	1.59
3.5	Blue	0.60
3.5	Black	0.78
3.5	Black	3.50
3.5	Green	1.04
7.5	Red	1.05
7.5	Red	1.78
7.5	Red	0.51
7.5	Black	0.99
7.5	Black	1.87
7.5	Black	2.57
7.5	Blue	2.49
7.5	Blue	1.00
11.5	Black	3.59
11.5	Black	4.03
11.5	Black	0.69
11.5	Blue	1.39
11.5	Blue	2.70
11.5	Blue	2.68
15.5	Blue	1.85
15.5	Blue	1.12
19.5	Black	3.32

19.5	Black	0.42
19.5	Black	0.66
23.5	Black	1.22
23.5	Blue	1.69

Robert Point – RP REF

Midpoint Depth (cm)	Fiber Colour	Microfiber Size
		Length (mm)
0.5	Black	0.86
0.5	Black	1.97
0.5	Black	1.42
0.5	Black	2.19
0.5	Black	1.00
0.5	Blue	0.54
0.5	Blue	0.97
0.5	Blue	1.02
0.5	Blue	1.89
3.5	Black	4.92
3.5	Black	1.28
3.5	Black	1.59
3.5	Black	0.31
3.5	Grey	1.49
7.5	Black	1.69
7.5	Black	3.12
7.5	Black	0.77
7.5	Blue	1.84
7.5	Blue	0.22
7.5	Blue	0.35
11.5	Red	1.59
11.5	Grey	1.14
11.5	Blue	0.88
11.5	Black	3.92
11.5	Black	3.85
11.5	Black	0.40
15.5	Black	4.10
15.5	Black	2.04

15.5	Black	1.39
15.5	Black	1.12
15.5	Blue	4.03
15.5	Blue	0.59
15.5	Blue	0.34
15.5	Green	2.51
19.5	Red	1.55
19.5	Black	4.98
19.5	Black	1.24
19.5	Blue	0.58
23.5	Green	1.49
23.5	Green	3.59
23.5	Black	2.58
23.5	Blue	0.99

Boundary Bay – T1MF1

Midpoint Depth (cm)	Fiber Colour	Microfiber Size
		Length (mm)
0.5	Black	1.06
0.5	Blue	2.22
0.5	Black	1.32
0.5	Black	3.95
3.5	Black	2.58
3.5	Black	0.84
3.5	Black	1.95
3.5	Black	2.05
3.5	Black	4.1
3.5	Black	1.58
3.5	Black	1.32
7.5	Red	0.99
7.5	Grey	2.86
7.5	Blue	3.58
7.5	Blue	0.537
7.5	Blue	0.731
7.5	Blue	0.876

7.5	Blue	2.89
11.5	Black	4.06
15.5	Blue	0.85
15.5	Blue	2.28
15.5	Blue	1.45
19.5	Black	1.03
19.5	Black	1.96
19.5	Blue	2

Boundary Bay – T2MF1

Midpoint Depth (cm)	Fiber Colour	Microfiber Size
		Length (mm)
0.5	Blue	2.38
0.5	Blue	1.9
0.5	Blue	0.52
0.5	Black	0.49
3.5	Blue	3.09
3.5	Blue	4
7.5	Grey	1.25
7.5	Blue	1.23
7.5	Blue	3.11
11.5	Grey	1.45
11.5	Black	2.04
15.5	Black	0.79
19.5	Green	0.88
19.5	Blue	2.63

Orchid Lake – C1-O

Midpoint Depth (cm)	Fiber Colour	Microfiber Size
		Length (mm)

0.5	Blue	0.63
0.5	Blue	0.24
0.5	Grey	0.81
3.5	Blue	0.32
7.5	N/A	N/A
11.5	N/A	N/A
15.5	Blue	0.73
19.5	N/A	N/A
23.5	N/A	N/A
27.5	Blue	0.36
31.5	Black	0.87
35.5	Black	0.61
35.5	Black	0.58
49.5	N/A	N/A
43.5	N/A	N/A

Appendix C. Laboratory Contamination Control

Date	Core	Type of Blank	Location	Microplastic Counts					
				Blue Fibers	Black Fibers	Grey	Red	Green	Total
05-Dec-18	T1MF1	Workstation	24 hours	3	1		1		5
05-Dec-18	T1MF1	Fume hood	24 hours	1	1				2
10-Dec-18	T2MF1	Workstation	24 hours	1	4				5
10-Dec-18	T2MF1	Fume hood	24 hours						0
03-Jun-19	C1-O	Workstation	24 hours	3	1				4
03-Jun-19	C1-O	Fume hood	24 hours	1					1
06-Jun-19	C1-O	Workstation	24 hours		1				1
06-Jun-19	C1-O	Fume hood	24 hours						0
18-Jun-19	C1-O	Workstation	24 hours		1				1
18-Jun-18	C1-O	Fume hood	24 hours	2				1	3
18-Jun-19	C1-O	Workstation	24 hours						0
15-Dec-19	GB REF	Filter paper	Fume hood	2	1				3
15-Dec-19	GB REF	Filter paper	Workstation	2	3				3
16-Dec-19	GB REF	Filter paper	Fume hood	1	1	1			3
16-Dec-19	GB REF	Filter paper	Workstation		3				3
17-Dec-19	GB REF	Filter paper	Fume hood			1	1		2
17-Dec-19	GB REF	Filter paper	Workstation		2				2
17-Dec-19	GB REF	Filter paper	Procedural blank		1				1
03-Jan-20	KC 3	Filter paper	Fume hood						0
03-Jan-20	KC 3	Filter paper	Workstation	1	2		1		4

03-Jan-20	KC 3	Filter paper	Procedural blank						0
04-Jan-20	KC 3	Filter paper	Fume hood		4				4
04-Jan-20	KC 3	Filter paper	Workstation	1	2			1	4
05-Jan-20	KC 3	Filter paper	Fume hood	2	1		1		4
05-Jan-20	KC 3	Filter paper	Workstation		1	1		1	3
07-Jan-20	RP REF	Filter paper	Fume hood	2	2				4
07-Jan-20	RP REF	Filter paper	Workstation	3	2				5
07-Jan-20	RP REF	Filter paper	Procedural blank		1				1
08-Jan-20	RP REF	Filter paper	Fume hood		3	1			4
08-Jan-20	RP REF	Filter paper	Workstation		2				2
08-Jan-20	RP REF	Filter paper	Procedural blank	1	2				3
09-Jan-20	RP REF	Filter paper	Fume hood	2	1				3
09-Jan-20	RP REF	Filter paper	Workstation		4		1		5
09-Jan-20	RP REF	Filter paper	Procedural blank		2				2

Appendix D. Microplastic Abundance

Grice Bay – GB REF

Sample ID	Midpoint Depth (cm)	Microplastic Counts						
		Blue Fibers	Black Fibers	Grey Fibers	Red Fibers	Green Fibers	Total	Total (corrected for contamination)
GB REF 0-1	0.5	2	7	2			11	8.43
GB REF 1-2	1.5	1	3	2			6	3.43
GB REF 2-3	2.5	2	6				8	5.43
GB REF 3-4	3.5		2			1	3	0.43
GB REF 4-5	4.5	1	3				4	1.43
GB REF 5-6	5.5	1	4				5	2.43
GB REF 6-7	6.5		6				6	3.43
GB REF 7-8	7.5		6				6	3.43
GB REF 8-9	8.5		2				2	0
GB REF 9-10	9.5	1	3				4	1.43
GB REF 10-11	10.5		4	1			5	2.43
GB REF 11-12	11.5		5				5	2.43
GB REF 12-13	12.5		3				3	0.43
GB REF 13-14	13.5		4		1	1	6	3.43
GB REF 14-15	14.5	1					1	0
GB REF 15-16	15.5		4	2			6	3.43
GB REF 16-17	16.5	2	3				5	2.43
GB REF 17-18	17.5		3				3	0.43
GB REF 18-19	18.5		2				2	0
GB REF 19-20	19.5		1				1	0
GB REF 20-21	20.5	3	2			2	7	4.43
GB REF 21-22	21.5	4	2				6	3.43
GB REF 22-23	22.5	1	2				3	0.43
GB REF 23-24	23.5	1	1		1		3	0.43
GB REF 24-25	24.5	1	1				2	0
GB REF 25-26	25.5	2	1				3	0.43

GB REF 26-27	26.5	2	1				3	0.43
GB REF 27-28	27.5		1				1	0

Kennedy Cove – KC3

Sample ID	**Midpoint Depth (cm)**	**Microplastic Counts**						
		Blue Fibers	**Black Fibers**	**Grey Fibers**	**Red Fibers**	**Green Fibers**	**Total**	**Total (corrected for contamination)**
KC3 0-1	0.5	2	4			1	7	4.06
KC3 1-2	1.5		2	2			4	0
KC3 2-3	2.5	2	1				3	0.06
KC3 3-4	3.5	1	2		1		4	1.06
KC3 4-5	4.5	1	3				4	1.06
KC3 5-6	5.5	2	3			2	7	4.06
KC3 6-7	6.5	1	3		1		5	2.06
KC3 7-8	7.5	2	3		3		8	5.06
KC3 8-9	8.5	2	3				5	2.06
KC3 9-10	9.5		3		1	1	5	2.06
KC3 10-11	10.5		4			2	6	3.06
KC3 11-12	11.5	3	3				6	3.06
KC3 12-13	12.5	3	1				4	1.06
KC3 13-14	13.5		2				2	0
KC3 14-15	14.5		1				1	0
KC3 15-16	15.5	2					2	0
KC3 16-17	16.5	1	4				5	2.06
KC3 17-18	17.5	1	1	1			3	0.06
KC3 18-19	18.5	1					1	0
KC3 19-20	19.5		3				3	0.06
KC3 20-21	20.5	1	3				4	1.06
KC3 21-22	21.5	2	1	1			4	1.06
KC3 22-23	22.5	1	3		1	1	6	3.06
KC3 23-24	23.5	1	1				2	0
KC3 24-25	24.5		1				1	0
KC3 25-26	25.5		4				4	1.06
KC3 26-27	26.5		2				2	0

Robert Point – RP REF

Sample ID	Midpoint Depth (cm)	Microplastic Counts						
		Blue Fibers	Black Fibers	Grey Fibers	Red Fibers	Green Fibers	Total	Total (corrected for contamination)
RP Ref 0-1	0.5	4	5				9	5.78
RP Ref 1-2	1.5		5			1	6	2.78
RP Ref 2-3	2.5		8		1	1	10	6.78
RP Ref 3-4	3.5		4	1			5	1.78
RP Ref 4-5	4.5	2	2				4	0.78
RP Ref 5-6	5.5		6	2	2		10	6.78
RP Ref 6-7	6.5	1	3	1			5	1.78
RP Ref 7-8	7.5	3	3				6	2.78
RP Ref 8-9	8.5	1	4	2			7	3.78
RP Ref 9-10	9.5	1	4	1			6	2.78
RP Ref 10-11	10.5	1	3			1	5	1.78
RP Ref 11-12	11.5	1	3	1	1		6	2.78
RP Ref 12-13	12.5	1	2			1	4	0.78
RP Ref 13-14	13.5		7	1			8	4.78
RP Ref 14-15	14.5	5	5	1		1	12	8.78
RP Ref 15-16	15.5	3	4			1	8	4.78
RP Ref 16-17	16.5	2	4	0			6	2.78
RP Ref 17-18	17.5	1	2	1			4	0.78
RP Ref 18-19	18.5	1	3		0		4	0.78
RP Ref 19-20	19.5	1	2		1		4	0.78
RP Ref 20-21	20.5	2	1				3	0
RP Ref 21-22	21.5	1	1				2	0
RP Ref 22-23	22.5			1		1	2	0
RP Ref 23-24	23.5	1	1			2	4	0.78
RP Ref 24-25	24.5		1				1	0
RP Ref 25-26	25.5		1				1	0
RP Ref 26-27	26.5		1				1	0
RP Ref 27-28	27.5	1					1	0

Boundary Bay – T1MF1

Sample ID	Midpoint Depth (cm)	Microplastic Counts						
		Blue Fibers	Black Fibers	Grey Fibers	Red Fibers	Green Fibers	Total	Total (corrected for contamination)
T1MF1 0-1	0.5	3	1				4	0.5
T1MF1 1-2	1.5	1					1	0
T1MF1 2-3	2.5	3	4				7	3.5
T1MF1 3-4	3.5		7				7	3.5
T1MF1 4-5	4.5	1	2				3	0
T1MF1 5-6	5.5	4					4	0.5
T1MF1 6-7	6.5		3				3	0
T1MF1 7-8	7.5	5		1	1		7	3.5
T1MF1 8-9	8.5	4					4	0.5
T1MF1 9-10	9.5	2	2				4	0.5
T1MF1 10-11	10.5	1	2				3	0
T1MF1 11-12	11.5		1				1	0
T1MF1 12-13	12.5	1					1	0
T1MF1 13-14	13.5	2	2				4	0.5
T1MF1 14-15	14.5	3	1				4	0.5
T1MF1 15-16	15.5	3					3	0
T1MF1 16-17	16.5	3			1		4	0.5
T1MF1 17-18	17.5	1	2				3	0
T1MF1 18-19	18.5		1				1	0
T1MF1 19-20	19.5	1	2				3	0

Boundary Bay – T2MF1

Sample ID	Midpoint Depth (cm)	Microplastic Counts						
		Blue Fibers	Black Fibers	Grey Fibers	Red Fibers	Green Fibers	Total	Total (corrected for contamination)
T2MF1 0-1	0.5	3	1				4	1.5
T2MF1 1-2	1.5	6					6	3.5
T2MF1 2-3	2.5	3	1				4	1.5
T2MF1 3-4	3.5	2					2	0
T2MF1 4-5	4.5	2			1		3	0.5
T2MF1 5-6	5.5	4	1				5	2.5
T2MF1 6-7	6.5			1			1	0
T2MF1 7-8	7.5	1		2			3	0.5
T2MF1 8-9	8.5	2					2	0
T2MF1 9-10	9.5	3					3	0.5
T2MF1 10-11	10.5	1	4				5	2.5
T2MF1 11-12	11.5		1	1			2	0
T2MF1 12-13	12.5	1	5				6	3.5
T2MF1 13-14	13.5	4	4				8	5.5
T2MF1 14-15	14.5		1				1	0
T2MF1 15-16	15.5		1				1	0
T2MF1 16-17	16.5	1					1	0
T2MF1 17-18	17.5		1				1	0
T2MF1 18-19	18.5		1				1	0
T2MF1 19-20	19.5	1				1	2	0

Orchid Lake – C1-0

Sample Name	Midpoint Depth (cm)	Microplastic Counts						
		Blue Fibers	Black Fibers	Grey Fibers	Red Fibers	Green Fibers	Total	Total (Corrected for contamination)

C1-O 0-1	0.5	2		1			3	1.57
C1-O 1-2	1.5	2	1				3	1.57
C1-O 2-3	2.5	2	2	1			5	3.57
C1-O 3-4	3.5	1					1	0
C1-O 4-5	4.5						0	0
C1-O 5-6	5.5						0	0
C1-O 6-7	6.5	3	2	1			6	4.57
C1-O 7-8	7.5						0	0
C1-O 8-9	8.5						0	0
C1-O 9-10	9.5	1					1	0
C1-O 10-11	10.5						0	0
C1-O 11-12	11.5						0	0
C1-O 12-13	12.5	1					1	0
C1-O 13-14	13.5						0	0
C1-O 14-15	14.5						0	0
C1-O 15-16	15.5	1					1	0
C1-O 16-17	16.5	1					1	0
C1-O 17-18	17.5	2					2	0.57
C1-O 18-19	18.5						0	0
C1-O 19-20	19.5						0	0
C1-O 20-21	20.5						0	0
C1-O 21-22	21.5						0	0
C1-O 22-23	22.5						0	0

C1-O 23-24	23.5						0	0
C1-O 24-25	24.5						0	0
C1-O 25-26	25.5						0	0
C1-O 26-27	26.5						0	0
C1-O 27-28	27.5	1					1	0
C1-O 28-29	28.5						0	0
C1-O 29-30	29.5						0	0
C1-O 30-31	30.5						0	0
C1-O 31-32	31.5		1				1	0
C1-O 32-33	32.5						0	0
C1-O 33-34	33.5						0	0
C1-O 34-36.5	35						0	0
C1-O 36.5-37.5	37		2				2	0.57
C1-O 37.5-38.5	38	1					1	0
C1-O 38.5-39.5	39						0	0
C1-O 39.5-40.5	40						0	0
C1-O 40.5-41.5	41						0	0
C1-O 41.5-42.5	42						0	0
C1-O 42.5-43.5	43						0	0
C1-O 43.5-44.5	44						0	0
C1-O 44.5-45.5	45						0	0

Appendix E. Microplastic Concentrations

Grice Bay – GB REF

Sample ID	Midpoint Depth (cm)	Microplastic Concentration (#particles/g)	Microplastic Concentration (#particles/kg)
GB REF 0-1	0.5	0.84	841.49
GB REF 1-2	1.5	0.34	341.94
GB REF 2-3	2.5	0.54	538.90
GB REF 3-4	3.5	0.04	42.82
GB REF 4-5	4.5	0.14	140.20
GB REF 5-6	5.5	0.24	242.61
GB REF 6-7	6.5	0.34	344.90
GB REF 7-8	7.5	0.34	341.84
GB REF 8-9	8.5	0.00	0.00
GB REF 9-10	9.5	0.14	141.85
GB REF 10-11	10.5	0.24	241.91
GB REF 11-12	11.5	0.24	242.15
GB REF 12-13	12.5	0.04	42.61
GB REF 13-14	13.5	0.34	342.73
GB REF 14-15	14.5	0.00	0.00
GB REF 15-16	15.5	0.34	339.74
GB REF 16-17	16.5	0.24	239.13
GB REF 17-18	17.5	0.04	42.19
GB REF 18-19	18.5	0.00	0.00
GB REF 19-20	19.5	0.00	0.00
GB REF 20-21	20.5	0.44	441.67
GB REF 21-22	21.5	0.34	337.10
GB REF 22-23	22.5	0.04	42.78
GB REF 23-24	23.5	0.04	42.58
GB REF 24-25	24.5	0.00	0.00
GB REF 25-26	25.5	0.04	42.35
GB REF 26-27	26.5	0.04	42.88
GB REF 27-28	27.5	0.00	0.00

Kennedy Cove – KC3

Sample ID	Midpoint Depth (cm)	Microplastic Concentration (#particles/g)	Microplastic Concentration (#particles/kg)
KC3 0-1	0.5	0.40	398.59
KC3 1-2	1.5	0.00	0.00
KC3 2-3	2.5	0.01	5.95
KC3 3-4	3.5	0.11	105.59
KC3 4-5	4.5	0.11	106.38
KC3 5-6	5.5	0.40	397.14
KC3 6-7	6.5	0.21	205.90
KC3 7-8	7.5	0.50	501.04
KC3 8-9	8.5	0.21	206.64
KC3 9-10	9.5	0.21	205.86
KC3 10-11	10.5	0.31	305.51
KC3 11-12	11.5	0.31	305.42
KC3 12-13	12.5	0.11	105.26
KC3 13-14	13.5	0.00	0.00
KC3 14-15	14.5	0.00	0.00
KC3 15-16	15.5	0.00	0.00
KC3 16-17	16.5	0.20	204.63
KC3 17-18	17.5	0.01	5.95
KC3 18-19	18.5	0.00	0.00
KC3 19-20	19.5	0.01	6.00
KC3 20-21	20.5	0.11	105.73
KC3 21-22	21.5	0.11	106.43
KC3 22-23	22.5	0.30	304.87
KC3 23-24	23.5	0.00	0.00
KC3 24-25	24.5	0.00	0.00
KC3 25-26	25.5	0.11	105.12
KC3 26-27	26.5	0.00	0.00

Robert Point - RP REF

Sample ID	Midpoint Depth (cm)	Microplastic Concentration (#particles/g)	Microplastic Concentration (#particles/kg)
RP Ref 0-1	0.5	0.57	572.73
RP Ref 1-2	1.5	0.28	275.06
RP Ref 2-3	2.5	0.67	672.82
RP Ref 3-4	3.5	0.18	176.55

RP Ref 4-5	4.5	0.08	77.42
RP Ref 5-6	5.5	0.67	666.86
RP Ref 6-7	6.5	0.18	178.02
RP Ref 7-8	7.5	0.28	276.98
RP Ref 8-9	8.5	0.38	377.47
RP Ref 9-10	9.5	0.27	273.43
RP Ref 10-11	10.5	0.18	177.66
RP Ref 11-12	11.5	0.27	274.43
RP Ref 12-13	12.5	0.08	77.75
RP Ref 13-14	13.5	0.48	477.71
RP Ref 14-15	14.5	0.88	877.56
RP Ref 15-16	15.5	0.48	477.81
RP Ref 16-17	16.5	0.28	276.95
RP Ref 17-18	17.5	0.08	77.74
RP Ref 18-19	18.5	0.08	77.81
RP Ref 19-20	19.5	0.08	78.08
RP Ref 20-21	20.5	0.00	0.00
RP Ref 21-22	21.5	0.00	0.00
RP Ref 22-23	22.5	0.00	0.00
RP Ref 23-24	23.5	0.08	76.37
RP Ref 24-25	24.5	0.00	0.00
RP Ref 25-26	25.5	0.00	0.00
RP Ref 26-27	26.5	0.00	0.00
RP Ref 27-28	27.5	0.00	0.00

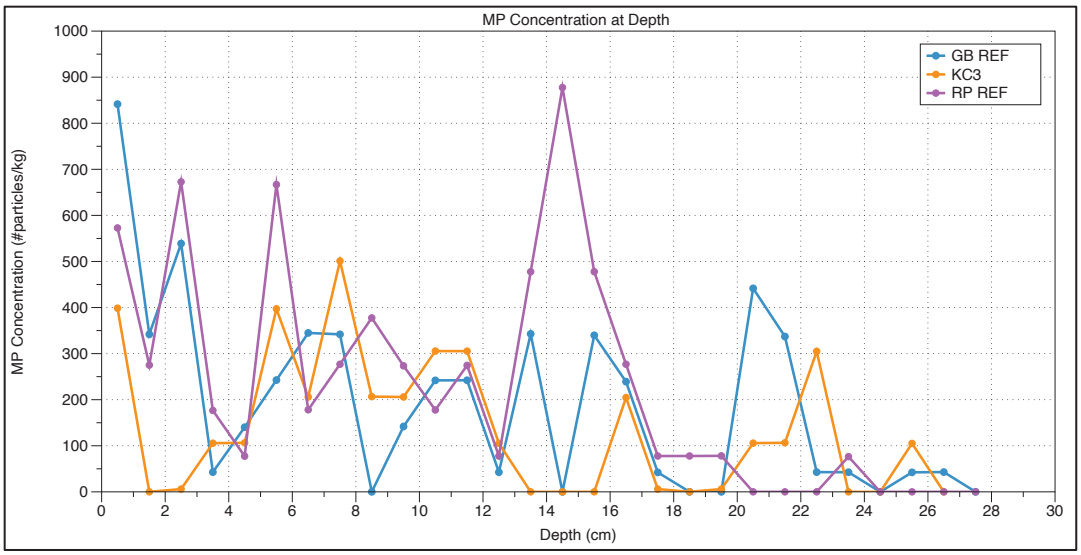


Figure E1. MP concentration (#particles/kg) at depth at Grice Bay (GB REF), Kennedy Cove (KC and Robert Point (RP REF)

Boundary Bay – T1MF1

Sample ID	Midpoint Depth (cm)	Microplastic Concentration (#particles/g)	Microplastic Concentration (#particles/ kg)
T1MF1 0-1	0.5	0.05	49.49
T1MF1 1-2	1.5	0.00	0.00
T1MF1 2-3	2.5	0.35	348.95
T1MF1 3-4	3.5	0.35	347.08
T1MF1 4-5	4.5	0.00	0.00
T1MF1 5-6	5.5	0.05	49.79
T1MF1 6-7	6.5	0.00	0.00
T1MF1 7-8	7.5	0.35	348.29
T1MF1 8-9	8.5	0.05	49.98
T1MF1 9-10	9.5	0.05	49.70
T1MF1 10-11	10.5	0.00	0.00
T1MF1 11-12	11.5	0.00	0.00
T1MF1 12-13	12.5	0.00	0.00
T1MF1 13-14	13.5	0.05	49.99
T1MF1 14-15	14.5	0.05	49.56
T1MF1 15-16	15.5	0.00	0.00
T1MF1 16-17	16.5	0.05	49.89

T1MF1 17-18	17.5	0.00	0.00
T1MF1 18-19	18.5	0.00	0.00
T1MF1 19-20	19.5	0.00	0.00

Boundary Bay – T2MF1

Sample ID	Midpoint Depth (cm)	Microplastic Concentration (#particles/g)	Microplastic Concentration (#particles/kg)
T2MF1 0-1	0.5	0.15	149.21
T2MF1 1-2	1.5	0.35	347.57
T2MF1 2-3	2.5	0.15	149.43
T2MF1 3-4	3.5	0.00	0.00
T2MF1 4-5	4.5	0.05	49.58
T2MF1 5-6	5.5	0.25	247.67
T2MF1 6-7	6.5	0.00	0.00
T2MF1 7-8	7.5	0.05	49.73
T2MF1 8-9	8.5	0.00	0.00
T2MF1 9-10	9.5	0.05	49.89
T2MF1 10-11	10.5	0.25	247.43
T2MF1 11-12	11.5	0.00	0.00
T2MF1 12-13	12.5	0.35	349.69
T2MF1 13-14	13.5	0.54	544.55
T2MF1 14-15	14.5	0.00	0.00
T2MF1 15-16	15.5	0.00	0.00
T2MF1 16-17	16.5	0.00	0.00
T2MF1 17-18	17.5	0.00	0.00
T2MF1 18-19	18.5	0.00	0.00
T2MF1 19-20	19.5	0.00	0.00

Orchid Lake C1-0

Sample ID	Midpoint Depth (cm)	Microplastic Concentration (#particles/g)	Microplastic Concentration (#particles/kg)
C1-O 0-1	0.5	0.31	311.88
C1-O 1-2	1.5	0.31	313.25
C1-O 2-3	2.5	0.70	697.67
C1-O 3-4	3.5	0.00	0.00
C1-O 4-5	4.5	0.00	0.00
C1-O 5-6	5.5	0.00	0.00
C1-O 6-7	6.5	0.91	908.55
C1-O 7-8	7.5	0.00	0.00
C1-O 8-9	8.5	0.00	0.00
C1-O 9-10	9.5	0.00	0.00
C1-O 10-11	10.5	0.00	0.00
C1-O 11-12	11.5	0.00	0.00
C1-O 12-13	12.5	0.00	0.00
C1-O 13-14	13.5	0.00	0.00
C1-O 14-15	14.5	0.00	0.00
C1-O 15-16	15.5	0.00	0.00
C1-O 16-17	16.5	0.00	0.00
C1-O 17-18	17.5	0.11	111.74
C1-O 18-19	18.5	0.00	0.00
C1-O 19-20	19.5	0.00	0.00
C1-O 20-21	20.5	0.00	0.00
C1-O 21-22	21.5	0.00	0.00
C1-O 22-23	22.5	0.00	0.00
C1-O 23-24	23.5	0.00	0.00
C1-O 24-25	24.5	0.00	0.00
C1-O 25-26	25.5	0.00	0.00
C1-O 26-27	26.5	0.00	0.00
C1-O 27-28	27.5	0.00	0.00
C1-O 28-29	28.5	0.00	0.00
C1-O 29-30	29.5	0.00	0.00

C1-O 30-31	30.5	0.00	0.00
C1-O 31-32	31.5	0.00	0.00
C1-O 32-33	32.5	0.00	0.00
C1-O 33-34	33.5	0.00	0.00
C1-O 34-36.5	35	0.00	0.00
C1-O 36.5-37.5	37	0.11	109.72
C1-O 37.5-38.5	38	0.00	0.00
C1-O 38.5-39.5	39	0.00	0.00
C1-O 39.5-40.5	40	0.00	0.00
C1-O 40.5-41.5	41	0.00	0.00
C1-O 41.5-42.5	42	0.00	0.00
C1-O 42.5-43.5	43	0.00	0.00
C1-O 43.5-44.5	44	0.00	0.00
C1-O 44.5-45.5	45	0.00	0.00

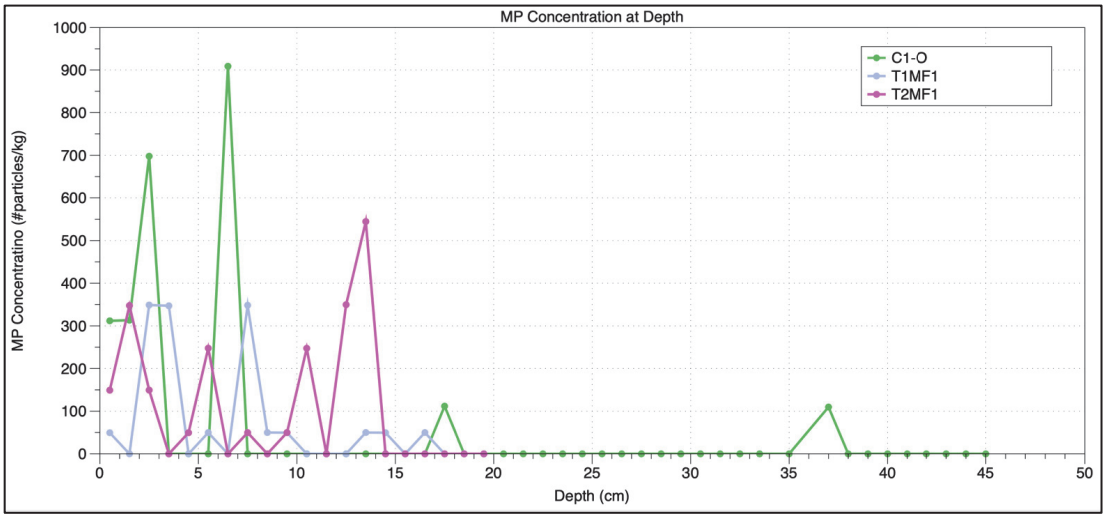


Figure E2. MP concentration (#particles/kg) at Orchid Lake (C1-O) and Boundary Bay (T1MF1, T2MF1).

Appendix F. Sediment Age, Sediment Mass Accumulation Rates (SMARs), and Microplastic Accumulation Rates (MPARs)

Grice Bay – GB REF

Sample ID	Midpoint Depth (cm)	Age in Years (CRS Model Estimate)	Year	Sediment Mass Accumulation Rate (g/cm^2/year)	Microplastic Accumulation Rate (#/cm^2/year)	Microplastic Accumulation Rate (#/100cm^2/year)
GB REF 0-1	0.5	3.05	2012.95	0.61	0.51	50.99
GB REF 1-2	1.5	6.38	2009.62	0.46	0.16	15.62
GB REF 2-3	2.5	9.73	2006.27	0.53	0.28	28.29
GB REF 3-4	3.5	13.18	2002.82	0.45	0.02	1.94
GB REF 4-5	4.5	16.57	1999.43	0.51	0.07	7.20
GB REF 5-6	5.5	20.07	1995.93	0.47	0.12	11.52
GB REF 6-7	6.5	23.62	1992.38	0.52	0.18	18.06
GB REF 7-8	7.5	27.34	1988.66	0.46	0.16	15.82
GB REF 8-9	8.5	31.02	1984.98	0.49	0.00	0.00
GB REF 9-10	9.5	34.97	1981.03	0.54	0.08	7.61
GB REF 10-11	10.5	38.93	1977.07	0.39	0.09	9.43
GB REF 11-12	11.5	43.30	1972.70	0.48	0.12	11.54
GB REF 12-13	12.5	47.66	1968.34	0.34	0.01	1.43
GB REF 13-14	13.5	52.64	1963.36	0.41	0.14	13.93
GB REF 14-15	14.5	57.62	1958.38	0.34	0.00	0.00
GB REF 15-16	15.5	63.52	1952.48	0.35	0.12	11.81
GB REF 16-17	16.5	69.42	1946.58	0.25	0.06	6.05
GB REF 17-18	17.5	76.93	1939.07	0.25	0.01	1.07
GB REF 18-19	18.5	84.44	1931.56	0.17	0.00	0.00
GB REF 19-20	19.5	93.91	1922.09	0.18	0.00	0.00
GB REF 20-21	20.5	103.38	1912.62	0.13	0.06	5.76

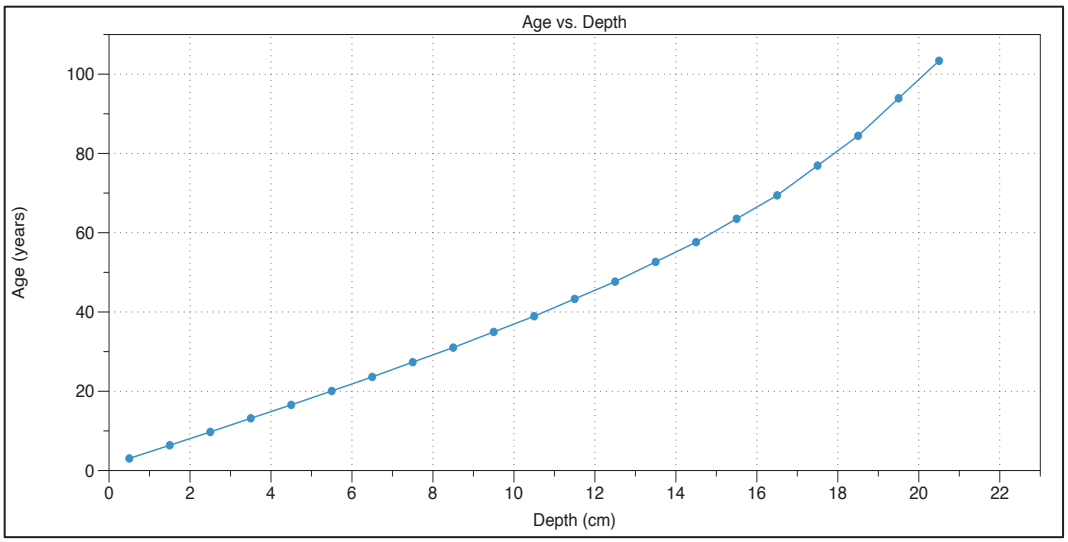


Figure F1. Age of sediment (CRS model) at depth at Grice Bay (GB REF).

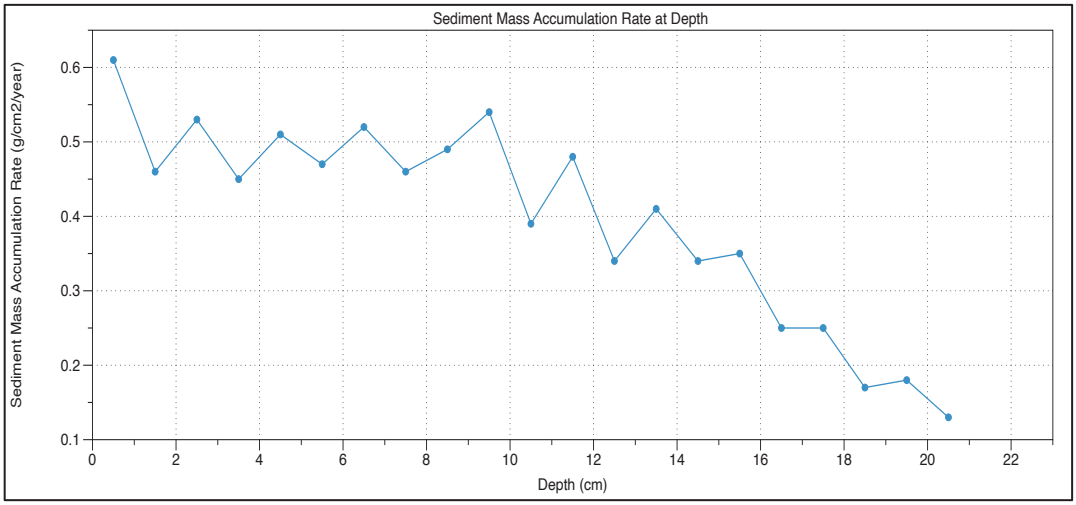


Figure F2. Sediment mass accumulation rate (g/cm2/year) at depth at Grice Bay (GB REF).

Kennedy Cove – KC3

Sample ID	Midpoint Depth (cm)	Age in Years (CRS Model Estimate)	Year	Sediment Mass Accumulation Rate ($g/cm^2/year$)	Microplastic Accumulation Rate ($\#/cm^2/year$)	Microplastic Accumulation Rate ($\#/100cm^2/year$)
KC3 0-1	0.5	3.00	2013.00	0.49	0.20	19.71
KC3 1-2	1.5	6.40	2009.60	0.46	0.00	0.00
KC3 2-3	2.5	10.00	2006.00	0.19	0.00	0.11
KC3 3-4	3.5	13.80	2002.20	0.36	0.04	3.77
KC3 4-5	4.5	17.20	1998.80	0.16	0.02	1.66
KC3 5-6	5.5	20.60	1995.40	0.27	0.11	10.75
KC3 6-7	6.5	23.80	1992.20	0.18	0.04	3.72
KC3 7-8	7.5	26.85	1989.15	0.49	0.25	24.61
KC3 8-9	8.5	29.90	1986.10	0.17	0.04	3.51
KC3 9-10	9.5	34.10	1981.90	0.20	0.04	4.06
KC3 10-11	10.5	38.30	1977.70	0.25	0.08	7.78
KC3 11-12	11.5	42.85	1973.15	0.28	0.09	8.60
KC3 12-13	12.5	47.40	1968.60	0.21	0.02	2.24
KC3 13-14	13.5	53.55	1962.45	0.24	0.00	0.00
KC3 14-15	14.5	59.70	1956.30	0.19	0.00	0.00
KC3 15-16	15.5	67.05	1948.95	0.18	0.00	0.00
KC3 16-17	16.5	74.40	1941.60	0.24	0.05	4.94
KC3 17-18	17.5	80.18	1935.83	0.12	0.00	0.07
KC3 18-19	18.5	93.30	1922.70	0.08	0.00	0.00
KC3 19-20	19.5	109.10	1906.90	0.11	0.00	0.06

KC3 20-21	20.5	124.90	1891.10	0.01	0.00	0.09

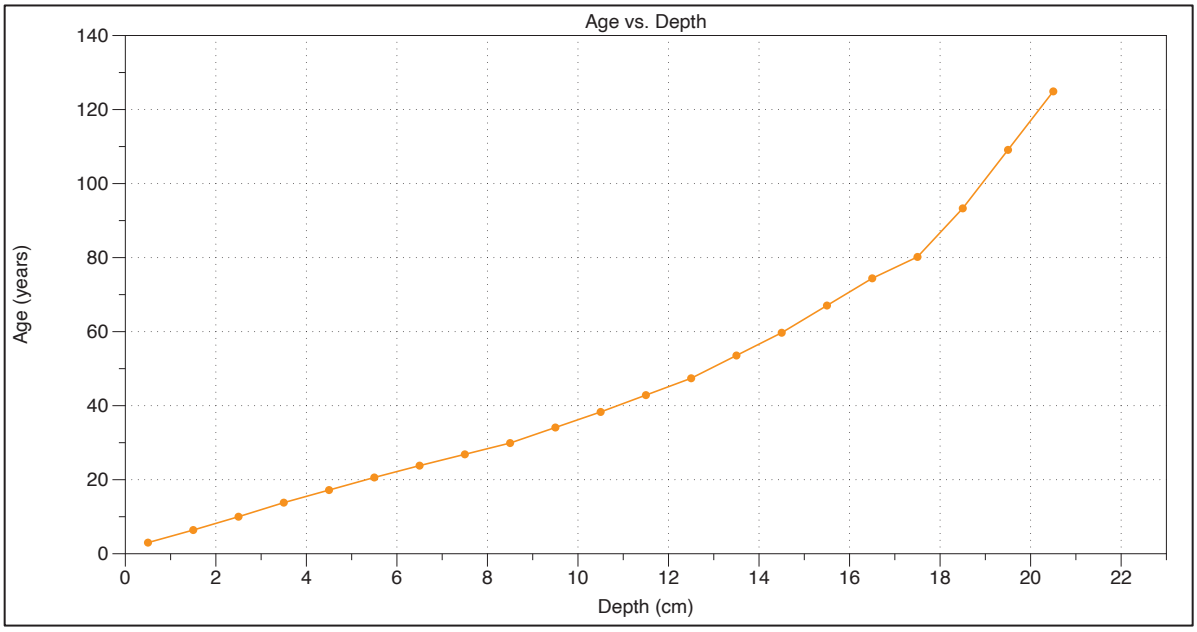


Figure F3. Age of sediment (CRS model) at depth at Kennedy Cove (KC3).

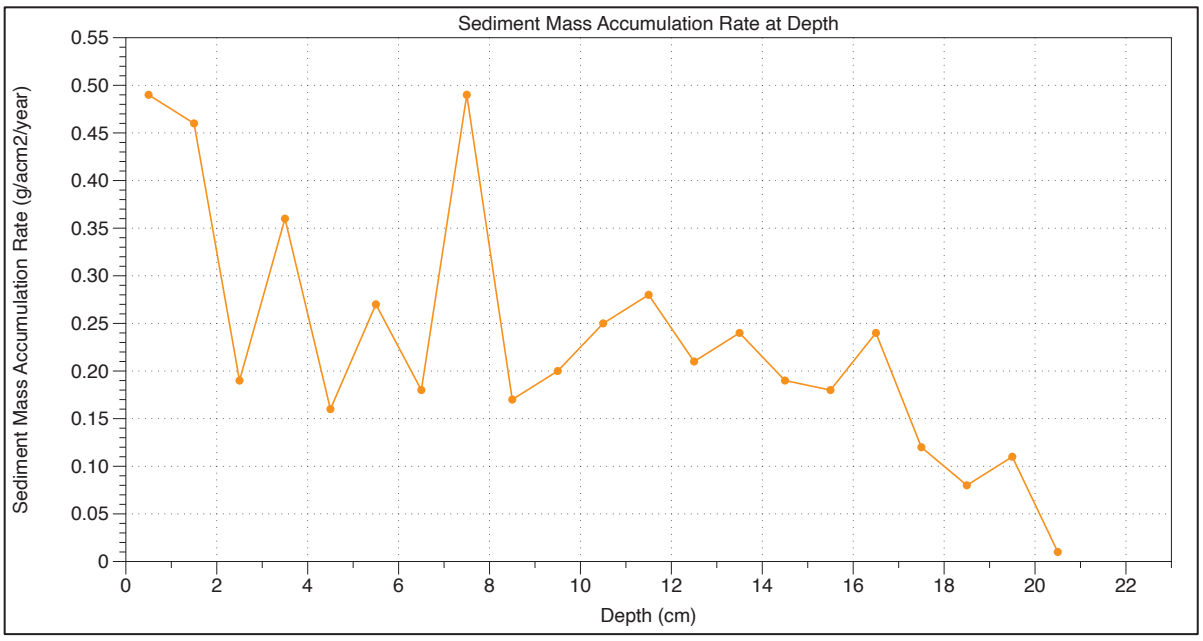


Figure F4. Sediment accumulation rate at depth at Kennedy Cove (KC3).

Robert Point – RP REF

Sample ID	Midpoint Depth (cm)	Age in Years (CRS Model Estimate)	Year	Sediment Mass Accumulation Rate (g/cm^2/year)	Microplastic Accumulation Rate (#/cm^2/year)	Microplastic Accumulation Rate (#/100cm^2/year)
RP REF 0-1	0.5	2.90	2013.10	0.49	0.28	28.03
RP REF 1-2	1.5	6.00	2010.00	0.68	0.19	18.82
RP REF 2-3	2.5	9.00	2007.00	0.50	0.33	33.47
RP REF 3-4	3.5	12.20	2003.80	0.74	0.13	13.01
RP REF 4-5	4.5	15.20	2000.80	0.49	0.04	3.81
RP REF 5-6	5.5	18.40	1997.60	0.54	0.36	36.22
RP REF 6-7	6.5	21.50	1994.50	0.36	0.06	6.46
RP REF 7-8	7.5	24.75	1991.25	0.72	0.20	19.88
RP REF 8-9	8.5	28.00	1988.00	0.27	0.10	10.01
RP REF 9-10	9.5	31.50	1984.50	0.27	0.07	7.46
RP REF 10-11	10.5	35.00	1981.00	0.54	0.10	9.68
RP REF 11-12	11.5	39.15	1976.85	0.49	0.13	13.47
RP REF 12-13	12.5	43.30	1972.70	0.23	0.02	1.77
RP REF 13-14	13.5	48.00	1968.00	0.32	0.15	15.46
RP REF 14-15	14.5	52.70	1963.30	0.27	0.23	23.42
RP REF 15-16	15.5	57.85	1958.15	0.35	0.17	16.64
RP REF 16-17	16.5	63.00	1953.00	0.20	0.05	5.43
RP REF 17-18	17.5	68.65	1947.35	0.31	0.02	2.38

RP REF 18-19	18.5	74.30	1941.70	0.03	0.00	0.22
RP REF 21-22	21.5	104.1	1911.90	0.02	0.00	0.00

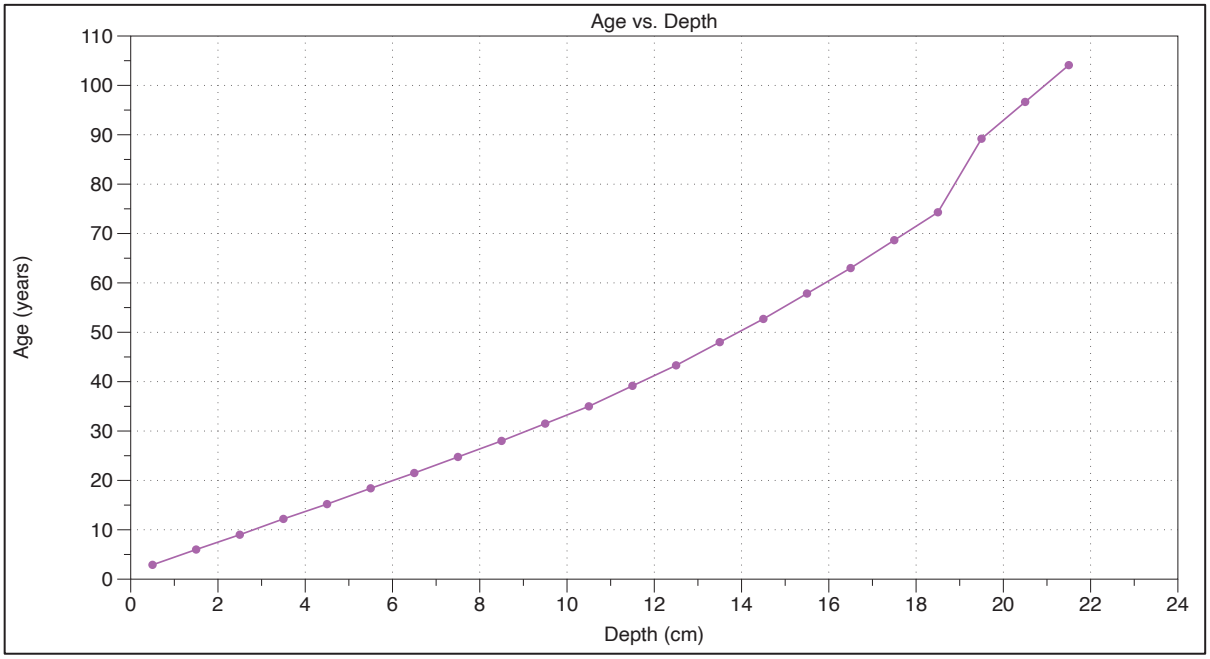


Figure F5. Age of sediment (CRS model) at depth at Robert Point (RP REF).

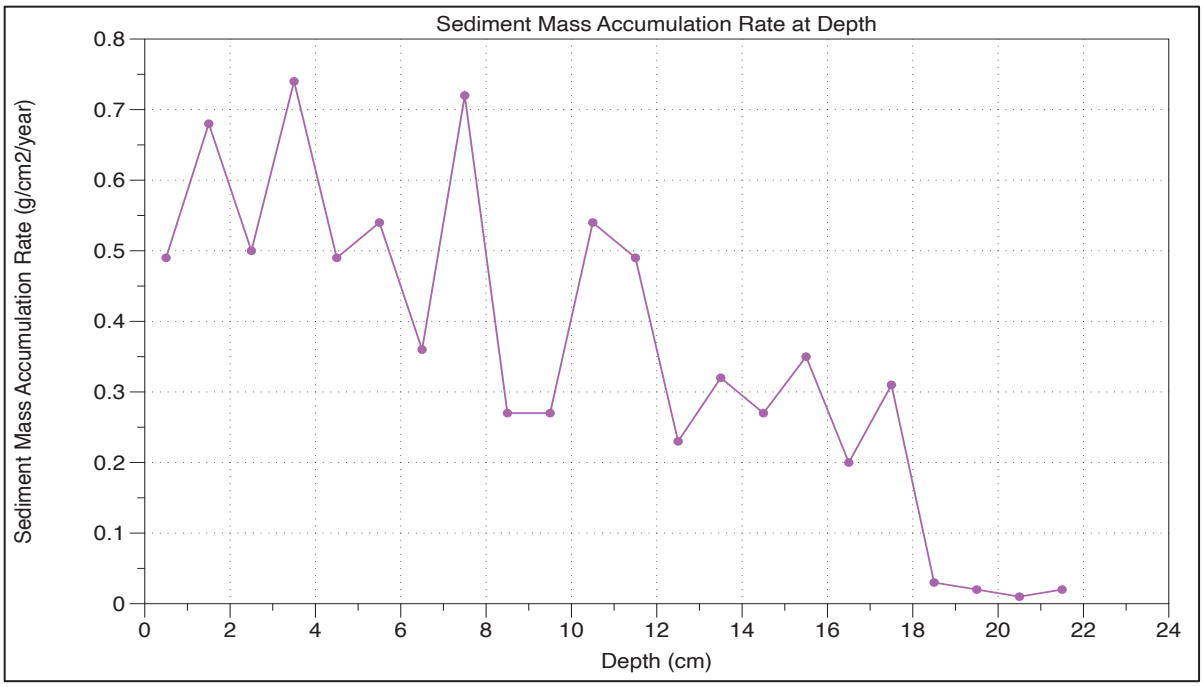


Figure F6. Sediment mass accumulation rate at depth at Robert Point (RP REF).

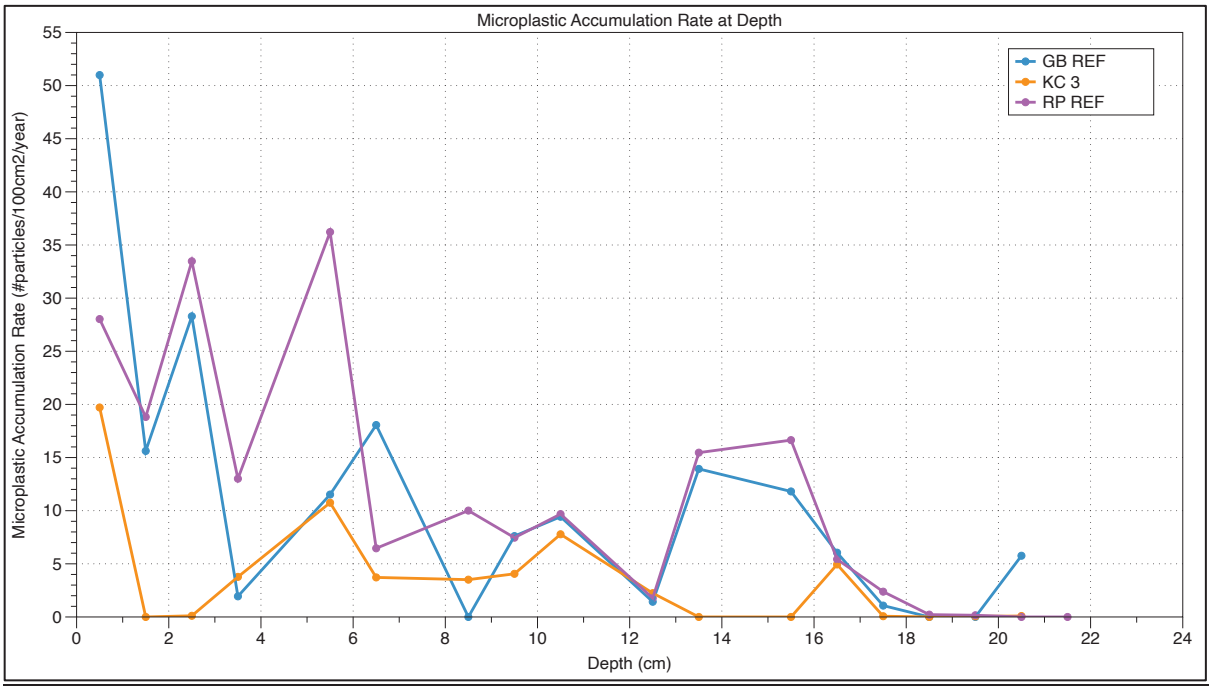


Figure F7. MPAR at depth at Grice Bay (GB REF), Kennedy Cove (KC3), and Robert Point (RP REF).

Appendix G. Sediment Grain Size Distribution

Grice Bay – GB REF

Midpoint Depth (cm)	Weight of Subsample (g)	>1 mm (g)	Weight % >1mm	0.5-1 mm (g)	Weight % 0.5-1mm	0.25-0.5 mm (g)	Weight % 0.25-0.5mm	0.125-0.25mm (g)	Weight % 0.125-0.25mm	0.0625-0.125mm (g)	Weight % 0.0625-0.125mm	<0.0625mm (g)	Weight % <0.0625
0.5	30.08	0.00	0.00	1.25	4.15	1.29	4.29	5.44	18.07	17.82	59.24	3.62	12.03
3.5	30.32	0.10	0.33	2.32	7.66	1.92	6.33	7.12	23.50	15.12	49.85	3.29	10.86
7.5	30.05	0.49	1.64	5.27	17.54	2.40	7.97	6.43	21.41	13.10	43.58	1.99	6.61
11.5	30.03	0.87	2.89	5.38	17.90	1.65	5.49	6.25	20.81	13.40	44.62	2.10	6.98
15.5	30.68	3.74	12.18	7.48	24.39	1.94	6.33	5.03	16.40	10.50	34.21	1.63	5.32
19.5	30.11	0.14	0.45	1.18	3.92	5.09	16.89	19.73	65.52	3.19	10.58	0.36	1.20
23.5	30.23	0.68	2.25	1.80	5.97	6.31	20.86	18.31	60.55	2.24	7.42	0.40	1.32
27.5	30.55	0.96	3.13	3.54	11.58	4.54	14.88	14.28	46.76	5.61	18.38	1.08	3.52

Kennedy Cove – KC3

Midpoint Depth (cm)	Weight of Subsample (g)	>1 mm (g)	Weight % >1mm	0.5-1 mm (g)	Weight % 0.5-1mm	0.25-0.5 mm (g)	Weight % 0.25-0.5mm	0.125-0.25mm (g)	Weight % 0.125-0.25mm	0.0625-0.125mm (g)	Weight % 0.0625-0.125mm	<0.0625mm (g)	Weight % <0.0625
0.5	30.11	21.90	72.74	3.24	10.77	1.45	4.82	2.05	6.82	1.06	3.50	0.32	1.06
3.5	29.95	10.46	34.91	8.35	27.87	6.27	20.94	3.90	13.01	0.86	2.88	0.11	0.38
7.5	29.91	6.72	22.47	6.66	22.27	5.18	17.31	7.48	25.00	3.34	11.18	0.53	1.78
11.5	30.18	4.40	14.57	7.64	25.32	6.14	20.36	8.01	26.53	3.13	10.36	0.86	2.86
15.5	28.04	2.38	8.48	6.26	22.32	2.51	8.94	8.42	30.03	10.09	35.97	0.77	2.74
19.5	29.29	7.77	26.53	7.57	25.84	3.82	13.06	6.33	21.62	3.32	11.33	0.48	1.63
23.5	29.25	5.29	18.07	7.17	24.52	8.01	27.36	4.29	14.67	3.47	11.88	1.02	3.50

Robert Point – RP REF

Midpoint Depth (cm)	Weight of Subsample (g)	>1 mm (g)	Weight % >1mm	0.5-1 mm (g)	Weight % 0.5-1mm	0.25-0.5 mm (g)	Weight % 0.25-0.5mm	0.125-0.25mm (g)	Weight % 0.125-0.25mm	0.0625-0.125mm (g)	Weight % 0.0625-0.125mm	<0.0625mm (g)	Weight % <0.0625
0.5	30.28	0.00	0.00	0.12	0.39	1.61	5.30	22.83	75.39	5.68	18.77	0.04	0.15
3.5	30.66	0.00	0.00	0.22	0.71	1.84	6.01	24.18	78.85	4.38	14.30	0.04	0.12

7.5	30.27	0.00	0.00	0.13	0.44	1.25	4.13	25.49	84.18	3.09	10.20	0.03	0.10
11.5	30.26	0.00	0.00	0.42	1.37	4.26	14.09	22.18	73.31	3.38	11.17	0.02	0.07
15.5	30.36	0.00	0.00	0.39	1.29	3.95	12.99	21.88	72.04	4.09	13.48	0.06	0.19
19.5	30.12	0.00	0.00	0.30	0.98	3.04	10.08	24.49	81.31	2.29	7.62	0.00	0.01
23.5	30.82	0.00	0.00	0.40	1.28	2.54	8.25	24.88	80.71	2.97	9.62	0.04	0.14
27.5	30.16	0.00	0.00	0.30	0.98	2.99	9.90	24.53	81.36	2.29	7.61	0.04	0.15

Boundary Bay – T1MF1

		Very Coarse Sand		Coarse Sand		Medium Sand		Fine Sand		Very Fine Sand		Silt and Clay	
Midpoint Depth (cm)	**Weight of Subsample (g)**	**>1 mm (g)**	**Weight % >1mm**	**0.5-1 mm (g)**	**Weight % 0.5-1mm**	**0.25-0.5 mm (g)**	**Weight % 0.25-0.5mm**	**0.125-0.25mm (g)**	**Weight % 0.125-0.25mm**	**0.0625 - 0.125 mm (g)**	**Weight % 0.0625 - 0.125 mm**	**<0.0625mm (g)**	**Weight % <0.0625**
0.5	30.28	0.00	0.00	0.70	2.33	2.70	8.91	19.66	64.92	6.71	22.14	0.51	1.70
3.5	30.88	0.00	0.00	0.38	1.22	2.79	9.03	20.98	67.95	6.17	19.97	0.56	1.82
7.5	30.00	0.00	0.00	0.80	2.66	9.85	32.83	16.07	53.56	3.12	10.39	0.15	0.49
11.5	29.88	0.00	0.00	0.32	1.07	1.99	6.67	20.41	68.31	6.78	22.68	0.38	1.27
15.5	29.87	0.00	0.00	0.54	1.82	5.66	18.94	18.50	61.95	4.86	16.28	0.30	1.02

19.5	29.77	0.00	0.00	0.92	3.09	1.18	3.96	20.14	67.67	7.16	24.06	0.36	1.22

Boundary Bay – T2MF1

		Very Coarse Sand		Coarse Sand		Medium Sand		Fine Sand		Very Fine Sand		Silt and Clay	
Midpoint Depth (cm)	**Weight of Subsample (g)**	**>1 mm (g)**	**Weight % >1mm**	**0.5-1 mm (g)**	**Weight % 0.5-1mm**	**0.25-0.5 mm (g)**	**Weight % 0.25-0.5mm**	**0.125-0.25mm (g)**	**Weight % 0.125-0.25mm**	**0.0625-0.125 mm (g)**	**Weight % 0.0625-0.125mm**	**<0.0625 mm (g)**	**Weight % <0.0625**
0.5	29.52	0.00	0.00	3.41	11.55	4.60	15.57	17.03	57.67	4.20	14.22	0.29	1.00
3.5	30.20	0.00	0.00	0.46	1.54	1.04	3.43	19.93	65.99	8.32	27.56	0.45	1.48
7.5	30.52	0.00	0.00	0.31	1.00	2.61	8.55	21.80	71.41	5.54	18.14	0.27	0.88
11.5	30.46	0.00	0.00	1.28	4.21	3.59	11.77	19.53	64.10	5.63	18.48	0.44	1.44
15.5	29.41	0.00	0.00	0.74	2.52	4.46	15.15	19.78	67.26	4.20	14.29	0.23	0.78
19.5	30.20	0.00	0.00	0.27	0.89	3.97	13.16	21.00	69.54	4.89	16.20	0.12	0.39

Orchid Lake – C1-0

		Very Coarse Sand		Coarse Sand		Medium Sand		Fine Sand		Very Fine Sand		Silt and Clay	
Midpoint Depth (cm)	Weight of Subsample (g)	>1 mm (g)	Weight % >1mm	0.5-1 mm (g)	Weight % 0.5-1mm	0.25-0.5 mm (g)	Weight % 0.25-0.5mm	0.125 - 0.25 mm (g)	Weight % 0.125-0.25mm	0.0625 - 0.125 mm (g)	Weight % 0.0625-0.125mm	<0.0625 mm (g)	Weight % <0.0625
0.5	5.06	0.00	0.00	0.00	0.00	0.00	0.00	0.05	0.95	0.14	2.71	4.31	85.20
3.5	5.03	0.00	0.00	0.00	0.00	0.00	0.00	0.19	3.68	0.20	4.00	4.65	92.33
7.5	5.16	0.00	0.00	0.00	0.00	0.04	0.78	0.10	1.88	0.59	11.40	4.44	85.97
11.5	5.04	0.00	0.00	0.00	0.00	0.02	0.46	0.15	3.02	0.42	8.31	4.45	88.29
15.5	5.12	0.00	0.00	0.00	0.00	0.01	0.10	0.28	5.49	0.52	10.18	4.31	84.18
19.5	5.13	0.00	0.00	0.00	0.00	0.07	1.31	0.22	4.27	0.49	9.57	4.35	84.78
23.5	5.10	0.00	0.00	0.00	0.00	0.08	1.49	0.20	3.82	0.50	9.83	4.28	84.02
27.5	5.00	0.00	0.00	0.00	0.00	0.01	0.16	0.20	3.90	0.40	7.98	4.28	85.71
31.5	5.10	0.00	0.00	0.00	0.00	0.10	1.90	0.10	2.04	0.68	13.39	4.21	82.67
35.5	5.01	0.00	0.00	0.00	0.00	0.08	1.66	0.29	5.81	0.63	12.67	4.00	79.86

Appendix H. Dry Bulk Density, $\%LOI_{550}$, $\%C_{org}$

Grice Bay – GB REF

Sample ID	Midpoint Depth (cm)	DBD (g/cm3)	$\%LOI_{550}$	$\%C_{org}$
GB REF 0-1	0.5	2.01	1.82	0.29
GB REF 1-2	1.5	1.53	1.94	0.32
GB REF 2-3	2.5	1.81	1.10	0.07
GB REF 3-4	3.5	1.54	1.64	0.23
GB REF 4-5	4.5	1.80	1.99	0.34
GB REF 5-6	5.5	1.69	0.81	0.00
GB REF 6-7	6.5	1.95	1.56	0.21
GB REF 7-8	7.5	1.70	1.66	0.24
GB REF 8-9	8.5	1.92	1.68	0.25
GB REF 9-10	9.5	2.12	1.33	0.14
GB REF 10-11	10.5	1.70	3.26	0.72
GB REF 11-12	11.5	2.08	1.44	0.18
GB REF 12-13	12.5	1.68	1.91	0.31
GB REF 13-14	13.5	2.02	1.68	0.25
GB REF 14-15	14.5	2.02	1.46	0.18
GB REF 15-16	15.5	2.05	1.33	0.14
GB REF 16-17	16.5	1.90	1.02	0.05
GB REF 17-18	17.5	1.91	1.29	0.13
GB REF 18-19	18.5	1.61	1.64	0.24
GB REF 19-20	19.5	1.75	1.48	0.19
GB REF 20-21	20.5	1.24	4.62	1.12
GB REF 21-22	21.5	1.22	1.26	0.12
GB REF 22-23	22.5	1.15	1.42	0.17
GB REF 23-24	23.5	1.11	1.49	0.19
GB REF 24-25	24.5	1.03	1.54	0.21
GB REF 25-26	25.5	1.48	1.47	0.18
GB REF 26-27	26.5	1.34	1.48	0.19
GB REF 27-28	27.5	1.14	1.55	0.21

Kennedy Cove – KC3

Sample ID	Midpoint Depth (cm)	DBD (g/cm3)	$\%LOI_{550}$	$\%C_{org}$
KC3 0-1	0.5	1.68	2.48	0.48
KC3 1-2	1.5	1.67	2.02	0.35
KC3 2-3	2.5	0.72	2.90	0.61
KC3 3-4	3.5	1.22	2.13	0.38
KC3 4-5	4.5	0.53	2.45	0.48
KC3 5-6	5.5	0.87	2.29	0.43
KC3 6-7	6.5	0.55	4.14	0.98
KC3 7-8	7.5	1.50	4.41	1.06
KC3 8-9	8.5	0.71	3.73	0.86
KC3 9-10	9.5	0.83	3.51	0.79
KC3 10-11	10.5	1.16	2.89	0.61
KC3 11-12	11.5	1.28	3.41	0.76
KC3 12-13	12.5	1.31	2.94	0.62
KC3 13-14	13.5	1.46	3.01	0.64
KC3 14-15	14.5	1.38	2.71	0.55
KC3 15-16	15.5	1.35	2.71	0.55
KC3 16-17	16.5	1.39	1.93	0.32
KC3 17-18	17.5	1.53	2.35	0.45
KC3 18-19	18.5	1.34	2.44	0.47
KC3 19-20	19.5	1.66	1.97	0.33
KC3 20-21	20.5	1.02	2.62	0.53
KC3 21-22	21.5	1.56	2.25	0.42
KC3 22-23	22.5	0.79	8.54	2.29

KC3 23-24	23.5	1.02	2.41	0.47
KC3 24-25	24.5	1.45	2.70	0.55
KC3 25-26	25.5	1.00	1.72	0.26
KC3 26-27	26.5	0.75	2.92	0.62

Robert Point – RP REF

Sample ID	Midpoint Depth (cm)	DBD (g/cm3)	$\%LOI_{550}$	$\%C_{org}$
RP Ref 0-1	0.50	1.52	1.85	0.30
RP Ref 1-2	1.50	2.05	0.89	0.01
RP Ref 2-3	2.50	1.59	0.92	0.02
RP Ref 3-4	3.50	2.21	1.20	0.10
RP Ref 4-5	4.50	1.58	1.02	0.05
RP Ref 5-6	5.50	1.68	1.29	0.13
RP Ref 6-7	6.50	1.18	1.09	0.07
RP Ref 7-8	7.50	2.33	0.95	0.03
RP Ref 8-9	8.50	0.93	1.35	0.15
RP Ref 9-10	9.50	0.95	0.80	0.00
RP Ref 10-11	10.50	2.26	0.79	0.00
RP Ref 11-12	11.50	2.04	0.97	0.04
RP Ref 12-13	12.50	1.07	0.86	0.00
RP Ref 13-14	13.50	1.52	0.83	0.00
RP Ref 14-15	14.50	1.37	0.87	0.00
RP Ref 15-16	15.50	1.79	0.83	0.00
RP Ref 16-17	16.50	1.11	0.75	0.00
RP Ref 17-18	17.50	1.73	0.78	0.00
RP Ref 18-19	18.50	2.13	1.01	0.05
RP Ref 19-20	19.50	1.83	0.97	0.03
RP Ref 20-21	20.50	1.37	0.90	0.01
RP Ref 21-22	21.50	1.82	1.38	0.16
RP Ref 22-23	22.50	1.34	0.82	0.00
RP Ref 23-24	23.50	1.62	1.20	0.10
RP Ref 24-25	24.50	1.79	1.24	0.12
RP Ref 25-26	25.50	1.85	0.89	0.01
RP Ref 26-27	26.50	1.65	0.82	0.00

RP Ref 27-28	27.50	2.02	1.16	0.09

Boundary Bay – T1MF1

Sample ID	Midpoint Depth (cm)	DBD (g/cm3)	$\%LOI_{550}$	$\%C_{org}$
T1MF1 0-1	0.5	0.89	1.63	0.23
T1MF1 1-2	1.5	0.93	2.48	0.49
T1MF1 2-3	2.5	0.97	1.41	0.17
T1MF1 3-4	3.5	0.92	0.50	0.00
T1MF1 4-5	4.5	0.90	1.05	0.06
T1MF1 5-6	5.5	1.15	1.05	0.06
T1MF1 6-7	6.5	0.88	0.73	0.00
T1MF1 7-8	7.5	0.96	1.39	0.16
T1MF1 8-9	8.5	1.04	0.87	0.00
T1MF1 9-10	9.5	0.93	0.84	0.00
T1MF1 10-11	10.5	0.97	0.15	0.00
T1MF1 11-12	11.5	0.94	1.33	0.14
T1MF1 12-13	12.5	0.86	1.11	0.08
T1MF1 13-14	13.5	1.00	0.79	0.00
T1MF1 14-15	14.5	0.95	1.49	0.19
T1MF1 15-16	15.5	1.06	0.99	0.04
T1MF1 16-17	16.5	1.02	0.84	0.00
T1MF1 17-18	17.5	0.93	3.12	0.68
T1MF1 18-19	18.5	1.00	0.55	0.00
T1MF1 19-20	19.5	0.94	3.44	0.77

Boundary Bay – T2MF1

Sample ID	Midpoint Depth (cm)	DBD (g/cm3)	$\%LOI_{550}$	$\%C_{org}$

T2MF1 0-1	0.5	0.92	1.35	0.15
T2MF1 1-2	1.5	0.89	1.20	0.10
T2MF1 2-3	2.5	0.88	1.32	0.14
T2MF1 3-4	3.5	0.90	1.22	0.11
T2MF1 4-5	4.5	0.88	1.27	0.12
T2MF1 5-6	5.5	0.91	0.86	0.00
T2MF1 6-7	6.5	0.88	1.37	0.15
T2MF1 7-8	7.5	0.90	1.45	0.18
T2MF1 8-9	8.5	0.93	1.53	0.20
T2MF1 9-10	9.5	1.07	1.29	0.13
T2MF1 10-11	10.5	0.98	1.54	0.21
T2MF1 11-12	11.5	1.01	1.97	0.33
T2MF1 12-13	12.5	1.13	1.57	0.21
T2MF1 13-14	13.5	0.98	1.19	0.10
T2MF1 14-15	14.5	0.99	1.45	0.18
T2MF1 15-16	15.5	0.97	1.11	0.07
T2MF1 16-17	16.5	1.02	1.45	0.18
T2MF1 17-18	17.5	0.97	1.49	0.19
T2MF1 18-19	18.5	1.14	1.36	0.15
T2MF1 19-20	19.5	1.06	1.87	0.30

Orchid Lake – C1-0

Sample ID	**Midpoint Depth (cm)**	**DBD (g/cm3)**	**%LOI_{550}**	**%C_{org}**
C1-O 0-1	0.5	0.57	22.64	10.46

C1-O 1-2	1.5	0.56	28.32	14.04
C1-O 2-3	2.5	0.56	28.17	13.95
C1-O 3-4	3.5	0.56	23.39	10.94
C1-O 4-5	4.5	0.57	23.28	10.87
C1-O 5-6	5.5	0.57	24.25	11.48
C1-O 6-7	6.5	0.57	28.05	13.87
C1-O 7-8	7.5	0.56	41.38	22.27
C1-O 8-9	8.5	0.58	29.07	14.51
C1-O 9-10	9.5	0.60	31.22	15.87
C1-O 10-11	10.5	0.57	45.25	24.71
C1-O 11-12	11.5	0.56	45.06	24.59
C1-O 12-13	12.5	0.57	34.48	17.92
C1-O 13-14	13.5	0.56	33.85	17.52
C1-O 14-15	14.5	0.55	29.76	14.95
C1-O 15-16	15.5	0.56	30.08	15.15
C1-O 16-17	16.5	0.61	76.45	44.36
C1-O 17-18	17.5	0.60	56.88	32.03
C1-O 18-19	18.5	0.57	29.80	14.97
C1-O 19-20	19.5	0.55	20.33	9.01

C1-O 20-21	20.5	0.56	21.07	9.47
C1-O 21-22	21.5	0.57	25.76	12.43
C1-O 22-23	22.5	0.61	33.75	17.46
C1-O 23-24	23.5	0.57	33.20	17.12
C1-O 24-25	24.5	0.57	25.14	12.04
C1-O 25-26	25.5	0.56	39.74	21.24
C1-O 26-27	26.5	0.58	37.08	19.56
C1-O 27-28	27.5	0.58	28.01	13.84
C1-O 28-29	28.5	0.60	35.67	18.67
C1-O 29-30	29.5	0.59	31.06	15.77
C1-O 30-31	30.5	0.61	19.27	8.34
C1-O 31-32	31.5	0.55	23.08	10.74
C1-O 32-33	32.5	0.56	29.86	15.01
C1-O 33-34	33.5	0.60	40.52	21.73
C1-O 34-36.5	35	0.59	52.43	29.23
C1-O 36.5-37.5	37	0.57	31.55	16.08
C1-O 37.5-38.5	38	0.60	31.99	16.36
C1-O 38.5-39.5	39	0.57	20.05	8.83

C1-O 39.5-40.5	40	0.56	27.89	13.77
C1-O 40.5-41.5	41	0.58	30.66	15.52
C1-O 41.5-42.5	42	0.57	25.72	12.41
C1-O 42.5-43.5	43	0.57	22.55	10.41
C1-O 43.5-44.5	44	0.60	40.04	21.42
C1-O 44.5-45.5	45	0.57	34.53	17.95

Appendix I. Microplastics in Marine Sedimentary Environments – Studies Used for Global Comparison (*** Indicate Studies Filtered Additionally For Contamination)

Location	Latitude, Longitude	Sedimentary Environment	Reported Mean Microplastic Concentration (# particles/kg)	Reported Median Microplastic Concentration (# particles/kg)	Types of particles reported	Sediment Depth (cm)	Coring Device	Key Observations	Reference
Lower Saxonian Wadden Sea ***	53.860300, 8.586836	Beach sediment	461		Fibers, granules	1	N/A	• Granules and fibers were present while fragments and polystyrene pellets were completely absent • No reported contamination	(Liebezeit & Dubaish, 2012)
Belgian Coast, Belgium ***	51.223670, 2.892331	Beach sediment	92.8 48.7-156.2		Fibers, granules, plastic films, polystyrene spheres	7	Van Veen Grab	• Particles found in all samples showing the wide distribution of microplastics in Belgian coastal waters • Highest concentrations in harbours • Depth profile of sediment cores suggested that microplastic	(Claessens et al., 2011)

								concentrations on the beaches reflect the global plastic production increase. • No reported contamination	
Halifax Harbour, Canada	44.605226, -63.430082	Beach sediment	5000 (2000-8000)		Fibers	4	Trowel	• Higher concentrations at the high tide line from the exposed beach and at the low tide line from the protected beaches • Reported contamination measures but no value	(Mathalon & Hill, 2014)
Southern Baltic Sea, Poland	54.517329, 18.788077	Beach sediment	39 ± 10	38	Fibers, plastic films, irregular fragments	2.5	Van Veen Grab	• MP concentrations lower in bottom sediments than in beach sediments • MP concentration decreased from the shore to the open, deep-sea regions	(Graca et al., 2017)

								• Polyester was the most common MP found • Polymer density an important factor influencing MP circulation with low density plastic debris probably recirculating between beach sediments and seawater in a greater extent than higher density • Consequently, low density MP concentration may be underestimated using current methods due to too small size of the debris • Contamination measures and no contamination of blanks	

Lesser Antilles, Caribbean	18.049799, -62.924392	Beach sediment	261 ± 6		Fibers	5	Metallic spoon	• >95% of MPs were fibers • Contamination measures, but no reported values	(Bosker et al., 2018)
East Coast, United States of America (Virginia/North Carolina) ***	35.989220, -75.509342	Beach sediment	1410 ± 810		Fibers, fragments	N/A	Corer (AMS metal split hand auger)	• MP particles detected in protected and unpolluted estuarine and beach sediments of Virginia and North Carolina • Contamination values reported (contamination range (mean ± standard deviation) was a 6 ± 2 fibers for blanks; for the Alcojet® solution, it was 17 ± 12 fibers; for the CsCl solution, it was 24 ± 9 fibers; and for the oil, it was 4 ± 2 fibers)	(Dodson et al., 2020)

								• Data not corrected	
Coast of Dubai, UAE	25.161831, 55.149724	Beach sediment	59.71		Fiber, string, fragment, polystyrene spheres	1	Stainless-steel spoon	• Blue fibrous MPS were dominant. • Contamination measures taken but no reported values	(Aslam et al., 2020)
Oman Sea, Oman	25.226351, 60.735406	Beach sediment	534 ± 26.8 (138.3 ± 4.5 to 930.3 ± 49.1)		Fiber, fragment, film, pellet	2	Stainless-steel shovel	• The major polymer was polyethylene, followed by polypropylene, and nylon • Fibers and fragments were dominant shapes of the MPs • Results confirmed the prevalence of the MPs as anthropogenic pollutants in the area • Measures taken, no reported contamination	(Kor et al., 2020)

Bay of Biscay, Spain	44.054024, -3.843658	Beach sediment	263.5 (145-382)		Fibers	5	N/A	• Open beaches contained a higher MP density than sheltered ones suggesting that many beached MP come from the ocean • Some measures taken but no reported contamination	(Masiá et al., 2019)
Coast of the Hengchun Peninsula, Taiwan ***	21.961485, 120.871145	Beach sediment	200		Fibers, fragments, pellets, films, foam	5	Metallic hand shovel	• There was no apparent seasonal difference but there were significant spatial differences among sampling sites • No significant difference in MP levels was observed among the west, south, and east coasts, but MP density was	(Chen & Chen, 2020)

								• higher on beaches with higher tourism activity levels • The most abundant type of MP was fibers (> 97%). • No Contamination values reported	
Eastern Harbour, Mediterranean Coast of Egypt	31.207672, 29.892304	Beach sediment	242		Fragments	5	N/A	• Baseline for MP research in Egypt • Utilized a new method of differential scanning calorimetry (DSC) to identify MPs • No contamination values reported	(Shabaka et al., 2019)
Aveiro, Portugal ***	40.627346, -8.750487	Beach sediment		100	Fibers and fragments	10	Metal shovel	• Major sources of MPs are tourism and marine traffic • Polyethylene, polypropylene, polystyrene,, nylon and polyester were	(Chouchene et al., 2021)

								• main MPs detected • No reported contamination	
Tanzanian Coastline 1. Mtoni Kijichi Creek 2. Mission Cross Beach 3. Ruvula ***	1. -6.840244, 39.300331 2. 6.432565, 38.905716 3. -10.395883, 40.435476	Beach sediment	1. 2972 ± 23 2. 589 ± 99 3. 15 ± 4		Fragments, fibers, and pellets	1	Metal spoon	• Polyethylene and polypropylene were the predominant MPs found • Marine protected areas had significantly less MPs than other areas along the coast • No reported contamination	(Mayoma et al., 2020)
Adriatic Sea, Slovenia ***	45.771949, 13.610541	Beach sediment	177		Fibers, fragments, and films	N/A	N/A	• Tourism did not seem to have an effect on macrodebris or microplastic quantity • No mention of contamination	(Laglbauer et al., 2014)
Mediterranean Sea, Santa Maria Marine Protected Area ***	39.507287, 2.412818	Shallow Coastal	270 ± 68 (100 ± 60 – 900 ± 10)		Fragments, filaments	3.5	Core tubes	• Sediments from Marine Protected Areas (MPAs) contained the highest	(Alomar et al., 2016)

								concentrations of MPs suggesting the transfer of MPs from source areas to endpoint areas • High proportion of MP filaments were found close to populated areas whereas fragment type MPs were more common in MPAs • There was no clear trend between sediment grain size and MP deposition in sediments • No reported contamination values	
Scapa Flow, Orkney, United Kingdom	58.885496, -3.081565	Shallow Coastal	2300		Fibers, particles	3	Metal-capped glass bijoux jar	• Detailed spatial and quantitative analysis revealed that MP distribution	(Blumenröder et al., 2017)

								was a function of proximity to populated areas and associated was tewater effluent . industrial install ations, degree of shore exposure and complex tidal flow patterns. • Contamination measures but no reported values	
Sekotong, Lombok-Indonesia	-8.741691, 115.904561	Shallow Coastal	48.3 ± 13.98		Fibers, fragments, granules, films	5-10	Stainless steel shovel	• Polystyrene the most abundant type of plastic polymer identified, followed by polyethylene and polypropylene. • Contamination measure but no reported values	(Cordova et al., 2018)
Guangdong Coastal Area, South China ***	21.601802, 112.279469	Shallow Coastal	(1444 ± 28) 433-4166		Fibers	5	Stainless steel shovel	• Transparent fibers were the main type of MP found in	(Yizheng Li et al., 2021)

								• both surface water and sediment • No reported contamination values	
Galway Bay, Ireland	53.236221, -9.072528	Shallow Coastal	60 ± 46		Fibers, fragments	5	Reineck box corer	• A relatively low abundance of MPs was found • All identified MPs were secondary • 99% fibers and 1% fragments • Polystyrene, polyester and polyethylene terephthalate • Reported contamination values (10 total)	(Pagter et al., 2020)
Chabahar Bay, Oman Sea, Iran	25.396148, 60.508772	Shallow Coastal	262 ± 17		Fibers, films, fragment	5	1m x 1m metal frame	• Most MPs were found at stations adjacent to urban, recreational, commercial and fishing sites • MPs mainly made of	(Kor et al., 2020)

								polyethylene and polyethylene terephthalate • Contamination assessed, and none found	
Red Sea Coast at Jeddah, Saudi Arabia	21.535896, 39.093802	Shallow Coastal	8 ± 7		Fibers, fragments, foams, and granules	1-10	Stainless steel shovel	• Polyethylene terephthalate and vinyl chloride vinyl-acetate copolymers were the dominant polymer types • Contamination assessed and no contamination found	(Al-Lihaibi et al., 2019)
Belgian Coast, Belgium ***	51.286800, 2.672495	Shallow Coastal	97 (48.7-156.2)		Fibers, granules, plastic films, polystyrene spheres	7	Van Veen Grab	• Particles found in all samples showing the wide distribution of microplastics in Belgian coastal waters • Highest concentrations in harbours	(Claessens et al., 2011)

								• Depth profile of sediment cores suggested that microplastic concentrations on the beaches reflect the global plastic production increase. • No reported contamination	
Adriatic Sea, Slovenia ***	45.624932, 13.541751	Shallow Coastal	155.6		Fibers, fragments, and films	N/A	N/A	• Tourism did not seem to have an effect on macrodebris or microplastic quantity • No mention of contamination	(Laglbauer et al., 2014)
Lower Saxonian Wadden Sea ***	53.859296, 8.433161	Shallow Coastal	11,600 (4,600-18,600)		Fibers, granules	1	N/A	• Granules and fibers were present while fragments and polystyrene pellets were completely absent • No reported contamination	(Liebezeit & Dubaish, 2012)
Hong Kong Harbour ***	22.288310, 114.170210	Shallow Coastal	164 (49-279)		Fragments, fibers, lines and pellets	N/A	Ekman dredge	• Identified microplastics include polypropylene,	(Tsang et al., 2017)

								low-density polyethylene, high density polyethylene • No mention of contamination	
Khark Island, Iran	29.290308, 50.330500	Shallow Coastal	690 (295-1085)		Fragments and fibers	5	Stainless steel spade	• Microplastics were detected in all sediment samples • A significant relationship exists between microplastics quantity and potentially toxic element/PAH concentrations • Contamination measures taken, no reported contamination	(Akhbarizadeh et al., 2017)
Gulf of Thailand	9.357888, 100.428945	Shallow Coastal	200 (90-320)		Films, sheets, fibers, fragments, beads	40	Gravity corer	• Microplastics composed of variety of polymers, including polyethylene (PE), polypropylene (PP), polystyrene	(Matsuguma et al., 2017)

								(PS), polyethylene-phthalates (PET), polyethylene-polypropylene copolymer (PEP), and polyacrylates (PAK) • Abundance of microplastics increased toward the surface, suggesting the global occurrence of and an increase in microplastic pollution over time • Contamination measures taken but no reported contamination	
Guadeloupe Island ***	16.175278, -61.537302	Shallow Coastal	215 (0-1040)		Fibers, fragments	N/A	Glass container	• Microplastics are found in all the studied sites and that their	(Sandre et al., 2019)

								distribution could be linked to marine currents or proximity to areas of significant human activities • No reported contamination or contamination measures	
Negros Oriental, Philippines	9.586111, 123.215239	Shallow Coastal	82		Fibers	N/A	Metal cylinder	• Microplastic particles extracted from subtidal sediment samples were dominated by semi-synthetic microfibers (rayon), probably from clothing and textiles • These microplastic types were absent in the guts of fish, likely due to the	(Bucol et al., 2020)

								different location and character of their feeding habitats • Contamination measured (2 microplastics detected)	
Singapore ***	1.270202, 103.893588	Estuarine	36.8 ± 23.6		Fibers, films, granules	3-4	Stainless steel spatula	• The majority of MPs were fibrous. A total of four polymer types were identified, including polyethylene, polypropylene, nylon and polyvinyl chloride. No relationship between sediment grain size and MP abundance found. • No contamination reported	(Mohamed Nor & Obbard, 2014)
Changjiang Estuary, China	31.523382, 121.448418	Estuarine	121 ± 9		Fibers, fragments, and pellets	5-10	Box corer	• No clear correlation between MPs	(Peng et al., 2017)

								and the finer sediment fraction was found. Rayon, polyester, and acrylic were the most abundant types of microplastics identified, indicating that the main source of microplastics in the Changjiang Estuary was from washing clothes. • Contamination reported (1 fiber)	
Guanabara Bay, Brazil	-22.838144, -43.156369	Estuarine	528 ± 30		Fibers, films, and fragments	~5	Van Veen Grab	• Guanabara Bay is amongst the most contaminated coastal systems worldwide by microplastics. Polyethylene and polypropylene	(Alves & Figueiredo, 2019)

								• most abundant polymers. • Contamination measured, no contamination in blanks	
Tampa Bay, Florida, United States of America	27.738542, -82.522224	Estuarine	280 ± 290		Fibers, fragments, flakes, beads, and films	N/A	Shipek grab sampler	• Highest concentrations of MPs were found in sediments close to industrial sources; lowest values in are consistent with shorter residence times. • Contamination measured and none found	(McEachern et al., 2019)
Jagir Estuary, Surabaya City, Indonesia ***	-7.120095, 112.656391	Estuarine	345 (92 – 590)		Fibers, films, and fragments	N/A	Ekman dredge sampler	• The MP shapes were comprised of fiber (57%), film (36%), and fragment (7%). The MP particles consisted of 68% large and 25% small	(Firdaus et al., 2020)

								sizes and comprised 56.7% polyester, 24.6% low-density polyethylene, and 18.8% polypropylene. The MP colors were 43% transparent, 21% black, 14% blue, 10% white, 8% red, and 4% yellow. • No contamination values reported	
Chongming Island, Yangtze Estuary, China (YRS Group) ***	31.529849, 121.600593	Estuarine	39.4 ± 16.1 (10-60)		Fibers and fragments	5-10	Sediment core sampler	• Particles identified were comprised of polyethylene, polypropylene, and a-cellulose and were mostly white and transparent • No contamination measured	(Yubo Li et al., 2020)

Jinjiang Estuary (Fujian, China) ***	24.849396, 118.648776	Estuarine	1926 ± 350.8 (980 ± 254.6 – 2340 ± 198)		Fibers, films, particle and fragments	0-3	Stainless steel shovel	• Restored regions of mangrove habitat showed higher concentrations of MPs than in mudflats, which points to mangrove restoration likely impacting MP deposition • Polyethylene terephthalate (PET), polyethylene (PE) and polypropylene (PP) were the main MPs found • No contamination measured	(Deng et al., 2020)
Muara Angke Wildlife Reserve, Indonesia	-6.104389, 106.767216	Estuarine	28.09 ± 10.28		Foams, fibers, fragments and granules	4-8	Stainless Steel Spatula	• Nearly 50% of MPs were polystyrenes which are associated with food packaging which is prone to degradation	(Cordova et al., 2021)

								• Polypropylene and polystyrene form the other 50%, which are polymers typically used in fishing and textiles • Measured contamination, none found	
Andong Salt Marsh, Hangzhou Bay, China ***	30.379696, 121.310782	Estuarine	264 ± 120		Fibers, fragments, films, and pellets	127.5-162.5	Columnar gravity sampler lined with a 60mm PVC core tube	• The main MP polymers were polystyrene, cellophane, polyethylene, and polypropylene • Procedural blanks run and 1.4 MPs/sample detected • No contamination protocol for airborne MPs	(Fraser et al., 2020)
Belgian Coast, Belgium ***	51.409920, 2.913632	Estuarine	170 (49-390)		Fibers, granules, plastic films, polystyrene spheres	7	Van Veen Grab	• Particles found in all samples showing the wide distribution of	(Claessens et al., 2011)

								microplastics in Belgian coastal waters • Highest concentrations in harbours • Depth profile of sediment cores suggested that microplastic concentrations on the beaches reflect the global plastic production increase. • No reported contamination	
Thames River, UK	51.492351, 0.522101	Estuarine	350 (185-660)		Fragments, fibers	10	Stainless steel scoop	• Microplastics were found at all four sites. • One site had significantly higher numbers of microplastics than other sites, 91% of which were fragments • At the remaining three sites, fibers	(Horton et al., 2017)

								were the dominant particle type • Contamination measured and 2 fibers per filter paper found (deemed negligible)	
Derwent Estuary, Tasmania	-42.886801, 147.352527	Estuarine	3315		Fibers, fragments, beads and sheets	N/A	N/A	• Microplastics observed in every sample, with greater plastic frequencies found in the upper (more recent) sediments • This time trend of microplastic accumulation matched that of global plastic production and coastal population growth. We observed that fibers were the most abundant type of microplastic in	(Willis et al., 2017)

								our samples. These fibers were present in sediments that settled prior to the presence of plastics in the environment • Contamination measures taken and adjustments for contamination made	
Warnow Estuary, Baltic Sea	54.124200, 12.092007	Estuarine	93 (46-100)		Fibers, fragments and spheres	10	Box corer	• Significant correlations between high-density polymer size fractions (≥500 µm) and sediment grain size • Sediment grain size was the critical proxy for MP abundance • Contamination measures taken	(Stolte et al., 2015)
Ombrone River Delta, Italy ***	42.672426, 11.023943	Estuarine	557 (45-1069)		Filaments, fragments and films	10-15	Manual bucket	• The investigated areas (rivers	(Guerranti et al., 2017)

								and shores) showed differences in amount of plastics present • Due to the presence of microplastics in coastal and transition areas, fluvial inputs may represent a significant plastic litter source and show considerable variations depending on factors related to the dynamics of the catchment • No mention of contamination	
Po River Delta, Italy ***	44.969389, 12.261574	Estuarine	20 (0-78)		N/A	5	Stainless steel shovel	• A hydrodynamic modelling approach was able to identify differing beaching rates	(Atwood et al., 2019)

								between various river mouths and suggested that particle beaching mostly occurred within the first 10 days of release • No contamination measures taken	
Baynes Sound, British Columbia, Canada ***	49.543062, -124.844354	Estuarine	6903 (540-25368)		N/A	23	Core	• Concentrations of microplastics determined within the intertidal sediment varied spatially and were similar to those found in other coastal regions of high urban use • Concentrations were independent of grain size and organic matter suggesting that	(Kazmiruk et al., 2018)

								physical processes other than those that govern natural sediment components determine the fate of microplastics within sediments • That high concentrations of microplastics have the potential to alter key sedimentary processes such as ammonium flu • Contamination not reported	
Venice, Italy	45.474674, 12.370886	Lagoon	1445		Fibers, fragments, films, and pellets	5	Box corer	• Higher concentrations generally observed in landward sites. Of the ten polymer types identified, the most abundant,	(Vianello et al., 2013)

								• accounting for more than 82% of total MPs, were polyethylene and polypropylene. • Contamination measured, none found	
North Tunisian Coast, Tunisia	37.398292, 10.063947	Lagoon	316.03 ± 123.74		Fibers, films, pellets, fragments, and Styrofoam	2-3	Stainless steel spatula	• Fibers significantly outnumbered plastic particles followed by fragments, Styrofoam, films and pellets. Except for industrial pellets, the presence of MPs is likely due to the degradation of marine plastic debris accumulating at each site. • Contamination measured, none found	(Abidli et al., 2018)

Halifax Harbour	44.664752, -63.430454	Lagoon	4100 (2200-6000)		Fibers	4	Trowel	• Higher concentrations at the high tide line from the exposed beach and at the low tide line from the protected beaches • Reported contamination measures but no value	(Mathalon & Hill, 2014b)
Yellow Sea, China	36.009248, 123.847186	Continental Shelf	97.8		Fibers, fragments, films, and pellets	N/A	Box corer	• MPs found in all samples, which emphasized their extensive distribution throughout the Bohai Sea and the Yellow Sea. Among the sampled MPs, fibers and small MPa were the most frequent types. • Contamination measured and none found	(Zhao et al., 2018)
Bohai Sea, China	38.929746, 119.777400	Continental Shelf	171.8		Fibers, fragments, films, and pellets	N/A	Box corer	• MPs found in all samples, which emphasized	(Zhao et al., 2018)

								their extensive distribution throughout the Bohai Sea and the Yellow Sea. Among the sampled MPs, fibers and small MPa were the most frequent types. • Contamination measured and none found	
Jiaozhou Bay, China	36.114013, 120.261054	Continental Shelf	25.0 ± 3.1	25.0	Fibers, fragments, films, and granules	45	PVC cylindrical sampler	• MP abundance was very low in deep sediment and generally showed a decreasing trend from the surface down with some shifts at different depths at certain sites. • Contamination measured but no values reported	(Zheng et al., 2020)
Mediterranean Continental Shelf, Spain	39.158698, 0.310092	Continental Shelf	113.2 ± 88.9		Fibers, fragments, films, and pellets	1.5	Stainless steel spatula	• Confirmed the widespread distribution of MPs in surface sediments from	(Filgueiras et al., 2019)

								the Spanish Mediterranean continental shelf • Measures taken, but no reported contamination values	
Argentinian Continental Shelf, Argentina	-39.276993, -61.727025	Continental Shelf	182.85 ± 115.14		Fibers	10	Shipek dredger	• First evidence of MP contamination at the Argentinean continental shelf. • Contamination measured and negligible	(Ronda et al., 2019)
Gulf of Thailand, Thailand	12.817281, 100.441407	Continental Shelf	150.4 ± 86.2		Fibers, granules, fragments and spheres	5	Stainless steel box sampler	• Secondary MPs are the dominant pollutant, while fibrous microplastics are mainly from municipal sewage discharge. • Contamination measures	(Wang et al., 2020)

								taken, no reported values	
Northern Tyrrhenian Sea ***	42.928696, 10.478818	Continental Shelf	1.70 ± 0.93		Fragment, filament, and film	N/A	Van Veen Grab	• Data represents a baseline for MP research in the Tyrrhenian seafloor. • No contamination measured	(Mistri et al., 2020)
1. Bering 2. Chukchi Seas	1. 63.520047, -176.508086 2. 68.495337, -171.475582	Continental Shelf	1. 17.69 2. 20.78		Fibers and films	5	Stainless steel box corer	• First records of MPs in surface sediment of Bering and Chuchki Seas • Primarily polypropylene followed by polyethylene terephthalate and rayon • Mostly fibers found • Contamination measured and reported • 1.85 ± 0.90 items/filter of microplastics was found in the	(Mu et al., 2019)

								• Blanks for the field samples, and 0.67 ± 0.58 items/filter were found in the procedural blanks for the laboratory samples	
Black Sea	44.203003, 32.031334	Continental Shelf	106.7		Fibers, films, fragments	5	Van veen grab and box corer	• Most MPs occurred in North-Western shelf • Mostly polyethylene and polypropylene found • Mostly black, blue and transparent fibers • Contamination measures taken but none reported	(Cincinelli et al., 2021)
Rias Baixas and Mino River Shelf Sediments (NW Spain)	42.404774, -8.969483	Continental Shelf	70.2 ± 74.2		Fibers, films, filaments, fragments, pellets, microbeads, paint sheets	5	Box corer dredge	• %TOC and sediment grain were measured and no relationship between MP concentrations	(Carretero et al., 2021)

								and %TOC or the smallest size fraction (<63um) of sediment grain size were found • Microplastic contamination measured and data corrected	
Southern Portuguese Shelf, Portugal	37.266162, -10.075582	Continental Shelf	10 ± 1		Pellets, fragments and fibers	N/A	N/A	• The vast majority were microfibers (25), identified as rayon fibres, and fragments (6) identified as polypropylene, through infrared spectroscopy • Contamination measures taken	(Frias et al., 2016)
North Sea, Belgium ***	51.599528, 2.538090	Continental Shelf	103 (72-270)		Fibers, granules, plastic films, polystyrene spheres	7	Van Veen Grab	• Particles found in all samples showing the wide distribution of microplastics in Belgian coastal waters	(Claessens et al., 2011)

								• Highest concentrations in harbours • Depth profile of sediment cores suggested that microplastic concentrations on the beaches reflect the global plastic production increase. • No reported contamination	
Southeast Baltic Sea	55.893167, 20.012172	Continental Shelf	34 ± 10		Fragments, fibers and films	N/A	Rectangular hand-operated dredge	• Microplastic particles were found in all of the samples with an average concentration of 34 ± 10 items/kg DW and have the same order of magnitude as neighbor studies reported • Contamination measures taken	(Zobkov & Esiukova, 2017)

Rothera Research Station	-66.841649, -68.882044	Continental Shelf	367		Fibers	3	Box core	• The highest concentrations of microplastic were recorded in sediment collected near the station sewage treatment plant outfall • Concentrations were similar to levels recorded in shallow and deep-sea marine sediments outside Antarctica. The detected microplastics had characteristics similar to those commonly produced by clothes washing • Contamination precautions taken	(Reed et al., 2018)
Rio de Janeiro Harbour ***	-23.012603, -43.151094	Continental Shelf	100		Fibers, films, fragments, pellets	N/A	Van veen grab	• The analyses detected the	(Baptista Neto et al., 2019)

								presence of MP in 100% of the samples, composed mainly by secondary microplastics, and almost 50% were fibers, followed by plastic films, plastic fragments and pellets • No contamination measures taken	
Mariana Trench ***	11.349144, 142.198836	Deep Sea	1600 (27-3900)		Fibers	N/A	N/A	• Results suggest that manmade plastics have contaminated the most remote and deepest places on the planet • The hadal zone is likely one of the largest sinks for microplastic debris on Earth,	(Peng et al., 2018)

								with unknown but potentially damaging impacts on this fragile ecosystem • No contamination measured mentioned	
Western Pacific Abyssal Plains	19.643660, 157.701272	Deep Sea	240 (0-1042)		Fibers, films, fragments	5	Stainless steel box core	• A significant correlation between the distribution of microplastics and the PCB concentrations in sediments was found • Microplastics were also detected in deep-sea organisms • This assessment of microplastics in deep-sea sediments and benthic organisms of the western Pacific Ocean	(D. Zhang et al., 2020)

								confirms that microplastic pollution exists in the deep-sea ecosystems of this region • Contamination measures taken	
Arctic Ocean	65.656274, -58.977050	Deep Sea	80		Fibers, fragments	2	Gravity and Piston Corers	• The discovery of microplastics in various environmental compartments of the Arctic Central Basin (ACB) suggested that these contaminants were potentially being transported to the deep-sea realm of this oceanic basin. For the first time, the present study conducted a preliminary assessment to determine	(Kanhai et al., 2019)

								whether microplastics were present in surficial sediments from the ACB • Contamination measures taken	
Rockall Trough	57.000654, -11.999743	Deep Sea	196 ± 129		Fibers, fragments, and films	60	OSIL Megacorer	• Significant negative trend was observed in the frequency of MPs with increasing sediment age, however there was an increase in polymer diversity. A number of mechanisms, including sediment reworking, could redistribute MPs vertically. Additionally, microplastics abundance	(Courtene-Jones et al., 2020)

								was significantly correlated with sediment porosity, suggesting interstitial transport via pore waters. • Contamination measured taken, but no values reported	
Hausgarten Observatory	78.990975, -1.529475	Deep Sea	4356 ± 675		N/A	5	Video-guided multiple corer (MUC; Oktopus GmbH)	• The northernmost stations had the highest quantities, indicating sea ice as a possible transport mechanism. The MP quantities are among the highest recorded from benthic sediments	(Bergmann et al., 2017)

								• Deep sea is a major sink for MPs • Contamination measured and samples blank corrected	

Appendix J. Microplastics in Marine Sedimentary Environments – Studies Not Included in Global Comparison

Location	Sedimentary Environment	Reported Mean Microplastic Concentration	Key Observations	Reason for Lack of Inclusion in Global Comparison	Reference
Southeastern Coastline South Africa	Beach sediment	688.9 ± 348.2 And 3308± 1449 particles/m^2	• No significant spatial patterns in either the sediment or water column MP densities, with little differences in density between bays and the open coast. • MPs were not associated with proximity to land-based sources or population density • governed by water circulation	Microplastics reported in particles/m^2	(Nel & Froneman, 2015)
Auckland, New Zealand	Beach sediment	459 particles/m^2	• High variability observed between the sites, indicating the importance of small-scale factors on MP contamination • Samples from high and intertidal zones showed no significant difference in MP contamination • MPs were predominately fibers (88%), with lower proportions of fragments (8%) and films (4%) • The majority of the microplastics analyzed were regenerated cellulose (34%), polyethylene terephthalate (22%) and polyethylene (15%)	Reported microplastics in particles/m^2	(Bridson et al., 2020)
Persian/Arabian Gulf	Beach sediment	N/A	• Considering the higher densities of PET, PVC, nylon, and polyester than seawater, they are likely to migrate downwards into marine sediments, a transfer process that has not been studied in detail.	Reported microplastics in particles/m^2	(Uddin et al., 2020)

Pearl River Estuary, Hong Kong	Beach sediment	5595 particles/m^2	• MP abundance significantly higher on the west coast than on the east coast, indicating that the Pearl River, which is west of Hong Kong, may be a potential source of plastic debris • Amounts of large plastic and microplastic debris of the same types (EPS and fragments) were positively correlated, suggesting that the fragmentation of large plastic material may increase the quantity of beach MP debris.	Reported microplastics in particles/m^2	(Fok & Cheung, 2015)
Gulf of Mexico Estuaries	Beach sediment/Estuarine	50.6 ± 9.96 particles/m^2 (marine dominated) and 13.2 ± 2.96 particles/m^2 (freshwater dominated)	• MPs ubiquitous throughout the area studied at concentrations 66-253 times larger than reported for the open ocean • Polypropylene and polyethylene were most abundant, with polystyrene, polyester and aliphatic polyamide also present but in lower quantities. • There was a gradient in MP abundance, with locations more directly exposed to marine currents and tides having higher MP abundance and diversity, as well as a higher contribution by denser polymers (e.g., polyester).	Reported microplastics in particles/m^2	(Wessel et al., 2016)
Southern Portuguese Shelf Waters, Portugal	Shallow Coastal	N/A	• MPs were found in 56% of sediment samples • The vast majority of MPs were microfibers	Did not report concentrations	(Frias et al., 2016)
Derwent Estuary, Tasmania	Estuarine	N/A	• Observed MPs in every sample, with greater plastic frequencies found in the upper (more recent) sediments.	No reported concentration values	(Willis et al., 2017)

			• This time trend of microplastic accumulation matched that of global plastic production and coastal population growth. • Fibers were the most abundant type of MP in samples • Fibers were present in sediments that settled prior to the presence of plastics in the environment		
South Carolina, USA	Estuarine	413.8 ± 76.7 particles/m^2 and 221.0 ± 25.6 particles/m^2	• MP concentration in these estuaries comparable to that reported for other estuaries worldwide • High abundance of black microplastic fragments believed to be tire wear particles	Reported microplastics in particles/m^2	(Gray et al., 2018)
Croatia	Continental Shelf	N/A	• All sediments contained plastics • Filaments were the most common shape category found • By weight, the most common polymer type was polyethylene, and by abundance the most common polymer type was	No reported concentrations	(Blašković et al., 2017)
Central Adriatic Sea	Shallow Coastal	310.81 particles/kg	• Fibers were the predominant MP found • No relationships observed between MP concentration and sediment grain size	Reported concentrations in m^2	(Mistri et al., 2017)
Kerkennah Archipelago, Tunisia	Shallow Coastal	611 particles/m^2	• Fibers and fragments were the most abundant • Most common MPs were polyethylene and polypropylene • MPs can act as vectors for trace metals	MPs reported in m^2	(Chouchene et al., 2020)
Algarve Coast	Deep sea	0-290 particles/kg	• Deep sea sediments may be an important sink for MPs	Oil extraction used	(Lechthaler et al., 2020)

Mukwei Lagoon	Lagoon	N/A	• MPs recorded in all samples • All samples revealed a peak of MPs around 10cm depth	No reported concentrations	(Chico-Ortiz et al., 2020)

Appendix K. Microplastic Accumulation Rates in Marine Sedimentary Environments

Site	Lat/Long	Environment Type	Depth/Year	Mean Concentration (# particles/g)	Mean Sediment Mass Accumulation Rate ($g/cm^2/year$)	Mean Microplastic Accumulation Rate (MPAR) (#particles/$100cm^2$/year)	Dating Method	Contamination Measures	Reference
Irish Continental Shelf	54.067153, -10.063250	Continental Shelf	1975-2015 (0-2cm)	No reported concentrations in volume or mass	N/A	N/A	AMS C^{14} dating	Contamination measures taken	(Martin et al., 2017)
Rockall Trough	57.000654, -11.999743	Deep Sea	1980-2020 (0-2cm)	0.215	0.07	1.50	^{210}Pb (CRS)	Contamination measures taken	(Courtene-Jones et al., 2020)
South China Sea	15.325469, 114.196824	Continental Shelf	1980-2017	0.111	N/A	N/A	^{210}Pb	Contamination measures taken	(Chen et al., 2020)
Tokyo, Japan	35.595337, 139.873249	Shallow Coastal	1971-2001	0.7	0.162	11.34	^{210}Pb (CRS)	Contamination measures taken	(Matsuguma et al., 2017)

Kuwait Bay, Kuwait	29.475572, 47.888750	Shallow Coastal	1980-2011	0.3	1.15	34.5	^{210}Pb (CS:CF) + ^{137}Cs	Contamination measures taken	(Uddin et al., 2021)
Andong Salt Marsh	30.379696, 121.310782	Shallow Coastal	N/A	0.22 ± 0.122	N/A	N/A	^{210}Pb (CIC)	Contamination measures taken	(Li et al., 2020)
Santa Barbara Basin	34.353838, -119.850029	Continental Shelf	1980-2010	N/A	N/A	28	Varve coupling	Samples corrected for contamination	(Brandon et al., 2020)
Clayoquot Sound	49.151055, -125.872147	Estuarine	1980-2016	0.288 ± 0.92	0.43	14	^{210}Pb (CRS)	Samples corrected for contamination	(Morra et al., 2021)