# Consumption Of Microplastics Impacts The Growth And Fecal Properties Of The Marine Copepod, Acartia Tonsa

CONSUMPTION OF MICROPLASTICS IMPACTS THE GROWTH AND FECAL PROPERTIES OF THE MARINE COPEPOD, *ACARTIA TONSA*

A Thesis Presented

by

Emily Shore

to

The Faculty of the Graduate College

of

The University of Vermont

In Partial Fulfillment of the Requirements
for the Degree of Master of Science
Specializing in Biology

May, 2020

Defense Date: March 19, 2020
Thesis Examination Committee:

Melissa H. Pespeni, Ph.D., Advisor
Joe Roman, Ph.D., Chairperson
Brent L. Lockwood, Ph.D.
Jason D. Stockwell, Ph.D.
Cynthia J. Forehand, Ph.D., Dean of the Graduate College

# ABSTRACT

Microplastics (<5 mm) are ubiquitous in the global environment and are increasingly recognized as a biological hazard, particularly in the oceans. Due to the small and pervasive nature of these particles, zooplankton have been known to consume and egest microplastics. Though zooplankton play critical roles in marine food webs and the biological pump through carbon rich fecal pellets, we know little about the effects of microplastics on early life stage growth, reproductive output, and carbon storage. Here, I investigated the effects of ingestion of low-density Polystyrene beads (5 $\mu$m) by the copepod *Acartia tonsa* on (1) early life stage (naupliar) growth, (2) adult fecundity and egg quality, (3) and fecal characteristics. I further explore potential impacts on carbon storage and the biological pump. *A. tonsa* were reared in one of two treatments: a 1:1 ratio of microplastics to 501 $\mu$g C $L^{-1}$ algae particles, or 501 $\mu$g C $L^{-1}$ algae only. Nauplii exposed to microplastics had shorter body lengths; additionally, adults produced eggs with smaller diameters and excreted smaller, more fragmented fecal pellets. Contaminated fecal pellet sinking rates were calculated to be 3.73 times slower and 2.29 times smaller than without microplastics. These two factors resulted in 8.56 times less fecal volume settling in the benthos per day for contaminated fecal pellets compared to control fecal pellets. Shorter zooplankton body lengths could reduce zooplankton morphology and population sizes, and thus impact higher trophic levels that depend on zooplankton as the critical link to the energy generated by primary producers. Slower fecal sinking rates can result in lingering carbon in the euphotic zone and increase the chance of contaminated fecal pellet consumption by coprophagous organisms. Taken together, these results suggest that microplastic consumption by zooplankton can impact the oceanic food chain, and slow carbon storage.

## ACKNOWLEDGEMENTS

I would like to express my deepest appreciation to my advisor Dr. Melissa Pespeni for her valuable advice, unparalleled support, and for being the biggest fan of microscopic fecal pellets. I would also like to extend my sincerest gratitude to Lauren Ashlock, for without her constant mentorship, encouragement, and support, I would not be where I am today. The completion of my thesis would not have been possible without the support and extensive knowledge of my committee members, and for that I am truly grateful. I am thankful for the rest of the Pespeni lab members for their continued support and joyful comradery we share in the workspace. I'd like to acknowledge the help of the undergraduate members of the Pespeni lab, for their behind-the-scenes help in animal care, and keeping the lab running smoothly. I must also thank the University of Vermont Biology Department for their funding support, Alicia Ebert for helping me create the beautiful and impactful images included in this thesis, and Christie Silkotch for her help as a wonderful science librarian. Last but certainly not least, I would like to thank my family and friends for their profound belief in my work, immense patience, and who never wavered in their support.

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# CHAPTER 1: COMPREHENSIVE LITERATURE REVIEW

## 1.1. Introduction

For decades, human activity has put immense stress on global marine ecosystems polluting the ocean with plastic debris. While the use of plastic products may seem like a critical part of our culture, large-scale production and heavy dependence on plastics only began in the early 1950s (Geyer et al., 2017). Since the start of industrial production, an estimated 8 million metric tons of plastic, including 236,000 tons of microplastics, make their way into the ocean annually (Jambeck et al., 2015). Plastic constitutes 10% of waste generated worldwide, and of this, 50% are single-use and non-renewable plastics (Mathalon & Hill, 2014) such as packaging materials, grocery bags, and straws (Cole et al., 2011). Marine plastic debris is an increasing biological and ecological issue due to their negative impacts on marine life (Alomar et al., 2016). The ocean produces over 50% of the world's oxygen and has high economic value through activities such as tourism and shipping (NOAA, 2020a). Marine ecosystems also provide critical sources of food for many human populations such as wild caught fish and natural plant commodities (Barbier, 2017). Thus, understanding the impacts of plastics on ocean ecosystems is vital.

A combination of large plastic debris, (macroplastics), and microplastics, (<5 mm in diameter) (Moore, 2008) make up 80 to 85% of all marine debris found either floating near the surface of the water column, or embedded in benthic sediment (Auta et al., 2017; Chiba et al., 2018; Jamieson et al., 2019; Law & Thompson, 2014). Macroplastics (>5 mm), such as plastic bags, commercial netting, and straws affect large marine biota such as whales (De Stephanis et al., 2013; Jacobsen et al., 2010; Unger et al., 2016) and economically important fish species (Miranda et al., 2016; Rochman et al., 2015). Due to the visible harm of these organisms, large oceanic plastic debris has been extensively researched since the early 2000s, such as in the North Pacific Gyre (Moore, 2008). Plastic consumption has been observed by at least 44% of marine bird species (Rios et al., 2007), and on beaches around the world (Corcoran et al., 2009; Le Guern, 2009). Less than 5% of all plastic material has been recovered and processed, leading to the vast accumulation of plastics in the marine environment (Sutherland et al., 2010). Although large plastic particles are easily viewed with the naked eye, oceanic plastic concentration predictions are still estimated with great uncertainty (Barnes et al., 2009; Jambeck et al., 2015). Fragmentation of larger plastics into microplastics has multiple components, including weathering, UV degradation, abrasion, and their combinations, over time (Hidalgo-Ruz et al., 2012; Thompson et al., 2004). These factors increase the abundance of marine litter throughout the water column, effectively increasing their availability to encounter rate with marine organisms (Law & Thompson, 2014). In addition, decreasing particle size increases the size range of species exposed, thus affecting a wide assortment of species (Botterell et al., 2019). Even though macroplastics have been the focus of study since the early 2000s, however both microplastics and large plastic debris can negatively affect marine life (Cole et al., 2013).

Microplastics are increasingly recognized as one of the new marine challenges in the Anthropocene. Microplastics are produced from a variety of plastic polymers, some chosen for their buoyancy properties, others for durability (Cole et al., 2011). Some common plastic types are polyvinyl chloride, nylons and polyethylene, polypropylene, and polystyrene (Auta et al., 2017). Plastics such as polyethylene and polypropylene are less dense than water, allowing them to float and resulting in high concentrations at the surface (SEA, 2012). Polystyrene is almost neutrally buoyant, allowing for the possible even distribution throughout the water column (Akester, 2019).

Polyvinyl chloride is negatively buoyant, concentrating these particles lower in the water column or settled in benthic sediment (Andrady, 2011; Jamieson et al., 2019). With this wide breadth of microplastic polymers, common quantification techniques of floating plastic debris seriously underestimate the amount of plastics in the ocean (Andrady, 2011). Consequently, removal and remediation methods can be complicated. Using surface-water collection such as shallow trawls usually excludes mid-water and benthic microplastics from concentration estimates (Andrady, 2011).

Microplastics enter and are carried throughout the ocean by many different processes. Transport vectors include Ekman currents (Kubota, 1994; Kubota et al., 2005; Martinez et al., 2009; Onink et al., 2019) stokes drift (Fraser et al., 2018; Onink et al., 2019), seasonal freezing/thawing of drifting Artic Sea ice (Bergmann et al., 2019; Geilfus et al., 2019; Lusher et al., 2015; Peeken et al., 2018) and windy/storm weather conditions (Lattin et al., 2004). The rate of degradation and persistence of these particles depends on type of polymer, density, shape, and durability (Eriksen et al., 2014). Most types of plastics are extremely durable, suggesting that the majority of polymers produced today will persist for at least decades, centuries, and possibly millennia (Hopewell et al., 2009).

Due to their small size and pervasive nature, microplastics have been ingested by marine organisms such as brown shrimp (*Crangon crangon*) (Devriese et al., 2015), amphipod crustaceans (Cau et al., 2019; Iannilli et al., 2018; Jamieson et al., 2019; Ugolini et al., 2013), barnacles (Goldstein & Goodwin, 2013), corals (Hall et al., 2015), oysters (Green, 2016), and zooplankton (Botterell et al., 2019; Cole et al., 2011, 2013, 2016, 2019; Coppock et al., 2019; Zhang et al., 2019a), however, the effects of microplastics on many other organisms have not been identified

(Nelms et al., 2018). These small marine organisms will consume microplastics by either false identification of food, or co-ingestion with algae, which can cause starvation and low fecundity for marine life (Cole et al., 2013, 2019; Ottvall & Carlsson, 2016). Small marine organisms provide critical ecosystem services, ranging from water filtration by oysters, storm barriers by corals, and zooplankton which play a key role in marine food webs (Botterell et al., 2019; Green, 2016; Hall et al., 2015). In the presence of microplastics, ecosystem services could be negatively affected, and the quality of marine environments could start to degrade. The full extent to which microplastic consumption impacts small oceanic organisms is ambiguous, but because both small plastic debris and zooplankton are similar in size, they are consistently found in equal amounts in surface trawls (Thompson et al., 2004). This strongly suggests that microplastics are in such high densities in the upper water column that they are readily available for zooplankton to ingest by means of non-selective filter feeding (Wirtz, 2012). With a non-selective filter feeding mechanism, all particles under a certain size are ingested by these small marine species (Kiørboe, 2011).

Zooplankton constitute a phenotypically and taxonomically diverse group (Pomerleau et al., 2015). They are distributed ubiquitously in the world's oceans, inhabiting surface waters to the benthic water column (McManus, 2012), and included in this group are copepods, which are the most abundant metazoan on earth (Turner, 2004). Zooplankton consume carbon that was fixed by planktonic phototrophs (algae) (Longhurst & Harrison, 1989), providing the critical link between primary producers and secondary consumers in the oceanic food web (Cole et al., 2019). Within the complex network of marine food webs, carbon and nutrients are circulated via feeding, fecal pellets, molts, carcasses, and vertical migration (Steinberg & Landry, 2017). Zooplankton, through these vectors, directly affect the elemental stoichiometry and material fluxes between particulate and dissolved matter, and in turn provide important drivers for the biological pump (Paffenhöfer et al., 2001). The biological pump, a combination of biological, chemical, and physical processes that sequester atmospheric $CO_2$ to the ocean's interior (Hain et al., 2013), is vital to slowing down the rate of climate change, absorbing about 30% of anthropogenic carbon emissions in the past decades (Bopp et al., 2020).

Copepod fecal material substantially contributes to the transfer of atmospheric carbon dioxide to the deep ocean via the biological pump (Coppock et al., 2019). Copepods readily incorporated polystyrene microspheres into their fecal pellets which decreased fecal sinking rates by 2.25-fold (Cole et al., 2016). Additionally, fecal pellets containing high density plastic polymers sank faster than pellets without microplastics (Coppock et al., 2019). The significant change of fecal sinking rate could prolong the amount of time microplastic ridden fecal pellets reside in the upper water column, where high concentrations of organisms are located. The decrease in sinking rates may also slow the rate of oceanic carbon storage, which, in the current climate crisis, can complicate the remediation efficiency of rising greenhouse gases (Folger, 2009). Factors already affecting plankton communities such as increasing ocean temperatures (Mackey et al., 2012; Richardson, 2008), decreasing oxygen (Marcus, 2011; Wishner et al., 2018), and ocean acidification (Hammill et al., 2018; Smith et al., 2016; Zervoudaki et al., 2015) are predicted to result in widespread changes in zooplankton carbon cycling in the future (Steinberg & Landry, 2017). By adding the effects of microplastics on zooplankton, their function in the biological pump may be hindered even more.

This review: 1) summarizes the current knowledge of the types of microplastics and their presence in the marine environment; 2) evaluates the effects of ingestion of microplastics on marine zooplankton with a focus on copepods; 3) suggests the potential bioaccumulation of microplastics in marine food webs; and 4) assesses zooplankton's role in the biological pump, and how the presence of microplastics affects the storage of oceanic carbon. It is important to understand the effects of microplastics will have on a foundational species found worldwide near the base of food webs, the copepod *Acartia tonsa*. This species has a wide geographical range, from the Atlantic to the Pacific Oceans, to coastal estuaries (Gonzalez, 2014).

## 1.2. Methods

During the months of December 2019 and January-March 2020, Web of Science was searched through the year 2020 with the following two terms denoted by the parentheses; "(Microplastic* AND zooplankton AND (ingest* OR "fecal pellets" OR "biogeochemical cycling" OR "biological pump" OR "marine food web" OR "feeding mechanism*" OR "oceanic food webs"))", and "(Microplastic* AND "marine plastic debris" OR "ocean gyres" OR "marine trophic transfer")". These searches resulted in 143 and 203 papers respectively. Each article title was reviewed and automatically eliminated if there was no relevance to the main themes of this review: the effects of microplastics on zooplankton, and oceanic microplastics. After this initial culling, remaining abstracts were read, and papers were further eliminated for the following reasons: 1) did not include microplastics, or if microplastic studies were done in extremely specific geographical areas i.e. remote marine reserves, 2) studies were conducted in freshwater, estuary, or terrestrial environments, 3) included species other than zooplankton (elimination did not apply to food web or trophic transfer studies), 4) focused on specific management or detection techniques, resulting in 184 papers included in this review. Historical papers, or papers with statistics and knowledge which may be considered outdated, were not automatically rejected.

These papers provide critical baseline microplastic information which is important to understand the growing problem of marine microplastic debris, including their effects on marine organisms.

## 1.3. Classes of Microplastics

Microplastics in the marine environment originate from a plethora of sources and vary greatly in particle size, shape, chemical composition and density (Duis & Coors, 2016). The release of microplastics occurs during all stages of the plastic life cycle, formulation, transport, usage, improper disposal such as industrial dumping, and erosion of larger particles (van Wezel et al., 2016). Plastic particles that are specifically manufactured to be microscopic in size, such as facial exfoliants and cosmetics, are classified as primary (Cole et al., 2011). Secondary microplastics result from the fragmentation of larger plastic products, such as discarded marine fishing equipment, bottles, and plastic bags, due to weathering and abrasion (Andrady, 2017). Both types of microplastic particles have numerous points of entry into the marine environment; however, due to their microscopic size, ascertaining exact concentrations or prolific sources is difficult. The following two sections aim to provide an overview of sources for both types of microplastic classifications.

### 1.3.1. Primary Microplastics

Primary microplastics are manufactured specifically to incorporate into cosmetic, industrial or medical products. Patented in 1972, plastic synthetic particles have been advertised and incorporated in facial cleansers, body washes, and soaps for a "deeper and better clean" (Beach, 1972). In addition to facial products, microplastics have been used in toothpaste, make-up, insect repellants, and baby products (Castañeda et al., 2014; Cole et al., 2011; Duis & Coors, 2016; Fendall & Sewell, 2009; Leslie, 2017). Up until the implementation of the 2015 Microbead-

Free Waters Act, primary microplastics took the place of natural products in traditional facial cleansers, such as pumice, ground walnut husks, and oatmeal (FDA, 2015; Fendall & Sewell, 2009). Small plastic pellets are produced as raw material, to manufacture larger plastic items such as plastic bags (Magnusson et al., 2016), and can wash into the ocean after accidental spilling during production, transport, or use (Andrady, 2017). These plastic pellets comprise common plastic polymers, such as polyethylene and polystyrene, which can leach chemical compounds in artificial sea water (Blastic, 2018). Primary microplastics are incorporated in blasting grit for sandblasting shipyards, gas exploration, fracking, painted metal constructions and other industrial uses (Sundt et al., 2014). Medically, these particles are incorporated in dentist tooth polish, medical disposables, medications, as well as antimicrobial agents (DG Environment Report, 2017).

### 1.3.2. Secondary Microplastics

A combination of physical, chemical, and biological processes reduces the structural integrity of large plastic debris, leading to fragmentation and formation of secondary microplastics (Browne et al., 2007). Discarded bottles, shoes, toothbrushes, fishing nets and ropes are just a few examples of larger plastic debris that serve as the 'parent material' for secondary microplastics (Cressey, 2016). Other sources include synthetic clothing; an estimated 700,000 fibers could fragment and be released from an average 6 kg load of acrylic fabric in household laundry (Napper & Thompson, 2016). Physical abrasion and turbulence by ocean waves in conjunction with direct exposure to sunlight on beaches and surface waters will degrade and fragment plastic rapidly (Barnes et al., 2009; Browne et al., 2007). The fragmented particles can either be integrated into beach sediment or washed back into the ocean (Wessel et al., 2016). Floating ocean plastic that is exposed to prolonged periods of sunlight can result in photo-degradation; ultraviolet (UV) radiation initiates photo-oxidative degradation (Andrady, 2011; Moore, 2008). Degradation can also occur thermooxidatively, without further exposure to UV radiation, allowing plastic debris that sinks below the ocean surface to possibly continue to fragment (Webb et al., 2013). Such processes are continual, with plastic debris breaking down further and further over time until they become ‘microplastic’ (<5 mm) in size (Cole et al., 2011; Moore, 2008).

### 1.4. Microplastics in the Ocean

Plastic debris encompasses a large size range, from visibly large and easily removable items, to microscopic particles that are difficult to filter out of sea water. Because of their size, microplastic detection and their adverse biological effects is challenging (Shim & Thomposon, 2015). The Manta trawl with a mesh size of 335 $\mu$m is used to collect and estimate microplastic concentrations, thus microplastics smaller than 335 $\mu$m will pass through the mesh undetected and will not be sampled (Botterell et al., 2019). This creates a large knowledge gap in small microplastic abundance and concentration in marine ecosystems. Despite current ocean cleanup efforts, such as public beach cleanups or ocean surface skimming, the number of plastic particles is likely to increase over time due to their durability and persistence (Shim & Thompson, 2015). To reduce oceanic plastic persistence, identifying original sources of macro and microplastics will help to provide prevention and remediation solutions. In this section, I identify sources of plastic litter of all sizes, as well as how microplastics may travel and converge in large masses in the water column. Considering the effects of both large and small plastic particles is important, for over the course of time, all will break down and fragment to microplastics.

### 1.4.1. Sources of Microplastics

Marine debris increases from improper disposal of plastic items such as incorrect recycling techniques or being dumped either directly or indirectly into the ocean (OSPAR, 2009). Of the total marine litter, 60 to 80% comprises plastics; this number can reach as much as 90 to 95% (Derraik, 2002). Microplastics can be characterized by their size, shape, color, chemical composition, and density, all of which create a variety of marine particles (reviewed by Andrady, 2017; Botterell et al., 2019; Cole et al., 2011; Hui et al., 2020; IUCN, 2015; Wright et al., 2013). Variation in plastic characteristics increases the number of marine ecosystems such as seagrass beds (Seng et al., 2020) and pelagic waters (Doyle et al., 2011), where these particles are found. Plastics in the marine environment come from two main sources: illegitimate dumping of garbage from ships at sea and land-based sources (Browne et al., 2007). Land-based sources include river-transported debris from farms, effluent from wastewater treatment plants, wind-transported litter, fibers resulting from clothes washing, and recreational plastic left on beaches (IUCN, 2018).

Land-based plastic debris contributes about 80% of the plastics in marine litter (Andrady, 2011). Microplastics present in cosmetics, intended to be washed down the drain, or synthetic fibers from laundry, an accidental byproduct, can enter the marine environment through industrial or domestic drainage systems (Auta et al., 2017; Miljødirektoratet, 2014). These systems lack sufficiently small sieves and filtration systems to remove small particles (Auta et al., 2017; Miljødirektoratet, 2014). When cosmetic microplastics were at their peak use in 2015, daily emissions from hand washing alone were 1 gram per capita, per year (Miljødirektoratet, 2014). Microplastics that are not removed from wastewater effluent can enter river systems or ground water tables which, during transport to the ocean, contaminate aquatic environments (Cole et al., 2011). For example, treatment plants with an estimated particle retention ability of 84% can still discharge up to 160 million microplastic particles a day in its effluent (Magni et al., 2019). Plastic particles that are retained and trapped in sewage sludge are often spread as fertilizer on agricultural lands, with the potential for human consumption (Mahon et al., 2017; Ryan et al., 2009). When sewage treatment plants become overloaded, they sometimes release untreated and unfiltered effluent into aquatic systems (Galafassi et al., 2019). Released effluent contaminated with microplastics can be further spread in aquatic systems by extreme weather such as flash flooding or hurricanes (Barnes et al., 2009).

Rainfall also has the potential to wash large amounts of microplastics into aquatic environments via runoff (Cole et al., 2011). Plastic pellets, or 'nurdles', are small particles that are industrially produced, and act as foundational virgin resin pellets to produce larger plastic pieces (Duis & Coors, 2016; Miljødirektoratet, 2014; Sundt et al., 2014). Resin pellets spilled during the manufacturing or transportation process can be transported or spread by rainfall into the marine environment (Weithmann et al., 2018). Also transported by rain, vehicle tires and road markings eventually erode and create particles that consist of numerous synthetic polymers (Grand View Research, 2020; Sundt et al., 2014). Studies in Denmark have reported that 2000 to 5600 tons of microplastics are discharged into effluent yearly from tires and textiles such as clothing (Lassen et al., 2015).

Around ~5 billion tons, or 60% of plastics manufactured to date, have been improperly discarded in the natural environment or are presently accumulating in landfills (Geyer et al., 2017). To reduce erosion, waste that is dumped in industrial countries' landfills is usually covered with soil or synthetic materials, and fences are implemented to reduce debris being blown away (Duis

& Coors, 2016). However, in developing regions and countries, the necessary infrastructure to deal with large quantities of waste is lacking, resulting in large quantities of plastics transported into the marine environment (Duis & Coors, 2016). Because the ocean is downstream from most human civilizations, lightweight waste lacking significant landfill infrastructure runs off into the ocean (Moore, 2008). When landfills become full, or reach their quota, debris is sometimes incinerated, shipped to other landfills, or allowed to overflow from storage warehouses, all of which have the potential to release plastic particles into the environment (Thompson, 2019). According to the Environmental Protection Agency (2016), the United States currently burns around 33 million tons of waste each year for energy generation, releasing microplastic particulates into the air. With landfills filling up around the world, they will increasingly add to the macro and microplastic issues in the marine environment (CalRecycle, 2018).

Far more abundant in earth's oceans are plastic particles typically derived from fragmentation of larger debris during use or weathering and degradation (Barnes et al., 2009). As previously decsribed, these particles constantly break off parent material into smaller and smaller pieces, exponentially adding to microplastic particulates on beaches, and the marine environment. Plastic debris on beaches have high oxygen availability, direct exposure to sunlight, and high temperatures, which cause rapid degradation through a series of chemical reactions (Andrady, 2011; Barnes et al., 2009). Pollution of marine and coastal environments with discarded and 'one use' plastic products is a rapidly increasing and significant global issue (UNEP, 2014). Debris collected during a beach cleanup along California's coast constituted of 54.3% recreation activities, 4.4% smoking, 3.2% ocean/waterway activities, and 0.5% medical/personal hygiene products (Thevenon et al., 2015). It is difficult to determine what percentage of this debris was directly deposited by beach goers (Moore, 2008), or if it washed onto the beach, however both sources can be traced back to human activity.

Coastal tourism, recreational and commercial fishing, marine vessels and marine industries (e.g. aquaculture, oil rigs) are maritime sources of plastic that can directly enter the marine environment (IUCN 2015, 2018). About 10% of the marine plastic debris found in the ocean can be attributed to the fishing industry (Thomas et al., 2019). When fishing nets become entangled on the bottom of the ocean, or when they break at sea, many fishing vessels choose to release the fishing nets into the ocean (Macfadyen, 2009). Discarded fishing nets have serious economic impacts such as entrapping and killing marine life or 'ghost fishing', as well as jamming boat propellers (NASA, 2010; NOAAb, 2020). Fishing nets and lines are commonly manufactured from synthetic materials and slowly degrade in the environment; when lost, fishing nets can drift thousands of kilometers into the open ocean (IUCN, 2015). These large fishing nets trap fish, seabirds, and larger marine biota such as whales and seals sometimes strangling and killing them (Hammer et al., 2012). Along with fishing vessels, commercial transport and recreational vessels contributed an estimated 6.5 million tons of plastic into the ocean in the early 1990s (Derraik, 2002) due to a lack of enforcement and education (OSPAR, 2009). During production or maintenance of these marine vessels, microplastic particles are used as alternatives to sandblasting for cleaning or smoothening vessel hulls (Derraik, 2002). Using microplastic particles for air blasting aids in cleaning rust and old paint off machinery and boat hulls (Pachkowski, 2016). The microbeads are supposed to be recycled to some extent, and collection methods exist on the market; however, general handling and collecting routines of this media is sometimes haphazard (Derraik, 2002).

### 1.4.2. Spatial Trends and Transport

Microplastics have been observed on beaches, floating on the sea surface, and embedded in the sea floor from coastal regions to the pelagic zone (Hidalgo-Ruz et al., 2012). With the wide dispersal of these particles, there are many global potential hazards for the marine environment. For example, colonization of plastic marine debris by sessile organisms provides a vector for transport of alien species in the ocean, such as invasive flora rafting on large plastic mats, which may threaten marine biodiversity (Gregory, 1991; Moore, 2008). Plastic mats can also leach chemicals into the nearby water column (Auta et al., 2017). Although microplastic particles have been discovered at oceanic depths of 10,890 meters, this area is extremely difficult to study due to its vastness which may add to the reasons why benthic sediment is not thoroughly studied at this time (Jamieson et al., 2019).

Driven by ocean currents, winds, river outflow and drift (Barnes et al., 2009; Onink et al., 2019), plastic debris can be transported vast distances to remote, otherwise pristine, locations (Cole et al., 2011). Microplastics contaminate shorelines on six continents from the poles to the equator, with higher accumulation in densely populated areas (Browne et al., 2011). The presence of microplastics can negatively impact the physio-chemical properties of the shoreline sediment, and less importantly perhaps, may be visually displeasing to onlookers (Cole et al., 2011). Net sampling does not collect smaller microplastics and no acceptable standard procedure is presently available for separating them from water or sand, thus exact concentrations remain unknown (Avio et al., 2017). The density of plastic debris on the surface of the ocean was considerably higher than benthic plastics after a storm event, which could increase the presence of coastal plastic debris (Lattin et al., 2004).

Microplastic concentrations were higher offshore compared to the coastal waters of California (Choy et al., 2019). This suggests that these particles may be transported by seasonally distinct winds and upwelling dynamics which varies per season (Choy et al., 2019). For microplastics, wind can play a significant role with low density plastic such as polystyrene, however for the transport of denser plastics, other vectors are at play (Chubarenko et al., 2016). Along with wind, the location of accumulation zones is largely determined by Ekman currents, which transport microplastics to subtropical gyres, and Stokes Drift which leads to increased plastic debris transport to the Arctic regions (Onink et al., 2019). An estimated average of 26,898 particles $km^{-2}$, ranging in size from 0.355 to over 4.750 mm, was found in the South Pacific subtropical gyre (Eriksen et al., 2013). In the western region of the South Pacific Subtropical Gyre, plastic pollution in an emerging contaminant on island shorelines and adjacent coastal waters (Eriksen et al., 2013), and in the southeast, aquaculture debris is the most significant debris contributor (Eriksen et al., 2013; Hinojosa & Thiel, 2009). Other garbage patches such as the "Eastern Garbage Patch", midway between Hawaii and San Francisco, is reportedly twice the size of the state of Texas and only continues to grow (The Great Pacific Garbage Patch, n.d.). These areas are sometimes called "deserts of the ocean," or regions with low nutrient levels (van Sebille, 2015). Caused by global change, alterations in sea surface temperature, patterns of precipitation, wind stress, severe storms, flooding events, and increasing sea levels will all make microplastics more available for transport in the seas (van Sebille, 2015).

### 1.5. Microplastics and Zooplankton

Due to their small and variable sizes, microplastics are ingested by numerous species ranging from microscopic marine invertebrates to large marine mammals (Nelms et al., 2019). As microplastic pollution and particle degradation continue to increase, their encounter rate with marine biota also increases. Microplastic particles have been documented to cause harm to aquatic organisms through both physical and chemical effects (Zhang et al., 2019b). Marine zooplankton are often found in waters with high microplastic concentrations; near the North Pacific Gyre, microplastics were six times more abundant than zooplankton (Seltenrich, 2015). After reviewing 43 studies looking at the effects of microplastics on fish, muscles, and zooplankton, Foley et al. (2018) suggested that zooplankton are among the most susceptible biota to microplastic exposure, as their growth, survival, and reproduction were negatively affected. The following five sections provide an overview of microplastics bioavailability, growth impediment, consumption, excretion, and survival for marine zooplankton, with a focus on copepods. Copepods are the most abundant organism with approximately 13,000 species, not only in the zooplankton community, but in the world's oceans (WoRMS, n.d.).

#### 1.5.1. Bioavailability

Microplastic particles are ubiquitous in the ocean. Recent estimates report the amount of circulating plastic particles in the ocean is reaching 5.25 trillion particles (Seltenrich, 2015). Several physical and biological factors can influence the bioavailability of microplastics to zooplankton, such as size, shape, age and abundance (reviewed by Botterell et al., 2019). Bioavailability is the key factor reflecting the potential toxicological influence of microplastics on different marine species (Ašmonaitė & Almroth, 2019). In a study of beach shorelines from sites across six continents, coastal plastic abundance to be highest in more densely populated areas (Browne et al., 2011). Near shorelines around the globe, there are many species of estuarine copepods, such as *Acartia tonsa*, which thrive in shallower waters and amongst these high plastic abundances. Remotely operated vehicles were used to examine the distribution of microplastics in the Monterey Bay pelagic environment and found particles in highest concentrations between 200 and 600 meters from the ocean's surface (Choy et al., 2019), where *A. tonsa* are known to inhabit (Gonzalez, 2014). These high particle concentrations near the shoreline or in ocean gyres is thought to be due to factors such as wind or ocean currents (Martinez et al., 2009).

High microplastic concentrations incorporated within marine organic debris blooms such as marine snow is due to the hydrophobic properties, or the tendency to repel away from water molecules, of plastics (Botterell et al., 2019). These aggregations amongst phytoplankton increase the likelihood of microplastic encounters with organisms, and consumption by zooplankton due to their non-selective feeding mechanisms (Long et al., 2015). Microplastics aggregate on the external surface of the dead copepod species *Temora longicornis* (Cole et al., 2013), which could lead to microplastic re-uptake by carnivorous zooplankton (Greve, 1977). Microplastics have also been trapped between the external appendages, such as swimming legs, feeding apparatuses, and antennae of live copepods, which can lead to reduced mobility and ingestion (Cole et al., 2013).

The quantification of microplastic concentrations in the marine environment is still uncertain, and current estimates could be low. This can be attributed to the vastness of the ocean, extreme depths (which are expensive and difficult to reach), constant mixing and circulating, as well as inadequate sampling mesh. Missing microplastics in tows may be due to too small of plastic pieces undetectable to conventional sampling nets, or because microplastics are sinking into the deep pelagic water column (Cózar et al., 2014). The scarcity of oceanic plastic debris data limits long-term studies, as well as monitoring of current plastic debris trends such as convergence zones and bioavailability to marine life, which could complicate remediation efforts (Morét- Ferguson et al., 2010).

### 1.5.2. Consumption

Microplastic ingestion has been observed in about 30 laboratory studies investigating marine holoplankton species, which are organisms that are planktonic for their entire lifecycle (Botterell et al., 2019; Rogers & Thorp, 2015). Consumption was detected in a wide variety of organisms and microplastic concentrations, from dinoflagellates sp. ($10^6$ microplastic particles $mL^{-1}$) (Hammer et al., 1999) to twin sailed salps (*Thetys vagina*, 2.23 microplastic particles $m^{-3}$) (Moore et al., 2001). Other examples include the copepod species *L. macrurus* and *Acartia* spp., which consumed microplastics when experimentally introduced; 43% of exposed *Acartia* spp. contained ingested microspheres after 3 hours (Setälä et al., 2014). Ingested microplastics were thought to be incorporated into the food-web via coprophagy and cannibalism (Setälä et al., 2014). The ecologically important cold-water copepod *Calanus finmarchicus* ingested nylon microplastic granules (10-30 $\mu$m), resulting in a 40% decrease in algal ingestion rates, and triggering premature molting in juvenile copepods (Cole et al., 2019). Microplastics also obstruct feeding appendages, reduce algal consumption, and stick to antennae which have mechanoreceptors that aid in prey detection (Cole et al., 2013). These findings all have the potential to reduce energy intake, as well as affect regular homeostatic functions in these marine invertebrates. The microplastic particle concentrations used in these experiments vary in environmental relevance, and some may be considered too high. Using a new autofluorescence method for detecting microplastics, a mean plastic concentration of 8277 particles $L^{-1}$ was found in California waters, averaging 5–7 orders of magnitude higher than previous studies (Brandon et al., 2020). This validates previous paper's justification for using high experimental microplastic concentrations, however, due to the complexities of extraction as well as the vastness of the ocean sampling, the full extent to which microplastics are ingested and can impact upon zooplankton is uncertain (Cole et al., 2013).

Microplastic consumption has also been studied in the field. Understanding all possible effects of microplastics on zooplankton is important, and when possible, studies in the natural environment provide insight on these questions. Five zooplankton groups were studied in the upper 200-m water column in the northern region of the South China Sea, where consumption of a wide range of microplastics was observed at all research stations (Sun et al., 2017). The highest abundance of the ingested microplastics was 103.49, 20.03, 2.83, and 5.16 pieces $m^{-3}$ for copepods, chaetognaths, jellyfish, and shrimps, respectively (Sun et al., 2017). Compared to the other organisms in this study, copepods had the highest percentage of individuals (79%) that ingested microplastics (Sun et al., 2017). In another region of the South China Sea off the coast of Malaysia, plankton tows were conducted and an average of one plastic particle was detected in 130 specimens, assayed from 6 groups of zooplankton (Md Amin et al., 2020). The average length of the ingested fibers and fragments were 534 +/- 372 $\mu$m and 61 +/- 12 $\mu$m respectively, which have been known to be severely under-assayed by other studies due to inadequate sampling technique (Md Amin et al., 2020). Microplastic abundances reported in this study were higher than those previously reported in Asia and other regions, which suggests that oceanic debris concentrations should be updated.

The wide range of zooplankton that have been found to ingest microplastics could be attributed to the variety of interspecies feeding mechanics. Kiøboe (2011) studied and summarized four major feeding types in zooplankton (1) passive ambush feeders: which passively encounter and intercept prey (2) active ambush feeders: which actively consume prey after a passive encounter (3) feeding-current feeders: which generate a feeding current and retrieve prey either by directly intercepting it, by filtering the prey out of the generated current, or by perceiving and capturing individual prey (4) and cruise feeders: which perceive and capture individual prey while cruising through the water. These feeding mechanisms allow for the organism to potentially misidentify microplastic particles for nutrients regularly ingested, possibly affecting filter feeders more than active feeders who are sometimes able to use complex behaviors to select between particles. With zooplankton filtering an hourly volume of sea water for prey particles $10^5$ times their own body volume, the potential for plastic ingestion is high (Jonsson & Tiselius, 1990). Because there are many different types of microplastic particles in the ocean, it is possible that marine organisms can misidentify these microplastics as their natural (Moore, 2008).

### 1.5.3. Fecundity

Marine copepods are the most numerous metazoan groups in the ocean (Reid & Williamson, 2010), supporting a large portion of secondary consumers, thus understanding effects on fecundity is important. During a two-generation exposure of *Tigriopus japonicus* to microplastics (0.23 mg $L^{-1}$), the number of nauplii per clutch and total fecundity for both generations significantly decreased (Zhang et al., 2019a). After a recovery period of one generation with no microplastics, the affected traits recovered indicating that microplastics do not display a transgenerational effect at the phenotypic level (Zhang et al., 2019a). However, if microplastics are found ubiquitously in the ocean, copepods may not have the opportunity to recover from plastic exposure. Prolonged exposure to 20 $\mu$m polystyrene beads (75 beads $mL^{-1}$) resulted in *Calanus helgolandicus* producing smaller eggs with reduced hatching success (Cole et al., 2015). In contrast, there was no significant effect of 'virgin' irregularly shaped polyethylene particles on *Acartia tonsa* egg production or egg hatching after a 48-hour exposure period, but the authors propose a more chronic exposure may reveal higher mortality (Bellas & Gil, 2020). Conversely, exposures of both microplastic particles and chlorpyrifos (0.1 $\mu$g $L^{-1}$), a broad-spectrum organophosphorus insecticide, egg production decreased 70% (Bellas & Gil, 2020). Due to its common use in both domestic and agricultural pest control, the presence of this insecticide is extensive and has been found sorbed to littoral plastic debris (León et al., 2018). This highlights the importance to study the combined effects of microplastics with other marine chemicals, as they are sometimes found in high concentrations together.

### 1.5.4. Growth and Survival

There are numerous studies investigating the effects of microplastic exposure on growth or survival of marine invertebrates, such as the onyx slippersnail (*Crepidula onyx*) (Lo & Chan, 2018) and *Daphnia magna* (Canniff & Hoang, 2018). However, studies including copepods are not as common. In order to grow and proliferate, maintain homeostasis, and create necessary lipid reserves, marine copepod *Calanus helgolandicus* required harnessed energy from their food sources (Cole et al., 2015). Impeded feeding mechanisms from the ingestion of microplastics can cause energy shortages and starvation, potentially stunting growth, and/or resulting in mortality in copepods. Juvenile copepods of the species *Calanus fimmarchicus* showed early molting when exposed to microplastics (Cole et al., 2019). It is thought that the reduced feeding, stymied lipid accumulation, or endocrine disruption caused by toxic compounds leaching from the particles may have contributed to the early molting (Cole et al., 2019).

After a two-generation exposure, microplastics significantly affected the copepod *Tigriopus japonicus* proteome, which translated to decreased survival and compromised reproduction (Zhang et al., 2019a). These impacts may have been caused by the high metabolic cost and reduced cellular energy stores from the consumption of environmentally relevant microplastic concentrations (Zhang et al., 2019a). In finding negative effects of microplastics in only a two-generation exposure, copepod species that have been living amongst microplastics for a long time may be seriously affected. Alternatively, marine zooplankton may have adapted to living in oceanic microplastic conditions. If not adapted, zooplankton growth and survival could be impacted, leading to population declines. Energy deficits from reduced algae intake and prolonged microplastic exposures experienced by early larval stages could have a detrimental effect on the growth and continued development to adulthood. In studying survival, adult *Calanus helgolandicus* copepods died during the exposure to 7-$\mu$m polystyrene microbeads, and their bodies were coated in microplastics (Cole et al., 2013). This finding could be attributed to the static or hydrophobic attractions between the experimental polystyrene beads, and the carbon rich carcasses (Cole et al., 2013). Some marine life can consume copepod carcasses, which could pose a problem if they are also ingesting large amounts of microplastics that may be adhered to carcasses (Greve, 1977).

### 1.5.5. Fecal Pellets

Zooplankton fecal pellets and their role in the ocean's biological pump has only recently been recognized as an important driver for carbon cycling. In the pelagic ocean, fecal pellets, an example of particulate organic matter (POM), transport photosynthetically produced fixed carbon to benthic sediment (Wieczorek et al., 2019). Organisms such as baleen and sperm whales (Roman et al., 2014), salps, a type of gelatinous zooplankton (Wieczorek et al., 2019), and copepods (Archibald et al., 2019; Cavan et al., 2019; see review Steinberg & Landry, 2017) have been studied to determine their role in driving the biological pump. Carbon removal from the atmosphere and sequestering to the sea floor is important because of the pace of global change and increasing atmospheric $CO_2$. Understanding this, it is vital to recognize the detriments microplastics have to this system, as they have been found to change fecal pellet characteristics and sinking rates. These changes could potentially affect the rate of carbon sequestering from the ocean's surface, as well as affect nutrient cycling to benthic organisms that depend on sinking POM (Cole et al., 2016).

There is a wide size range of plastic particles ingested and encapsulated in copepod fecal pellets. *Tigriopus japonicas* consumed and egested high concentrations of nano-sized particles (0.05 $\mu$m and 0.5 $\mu$m), displaying a preference towards the plastic beads over phytoplankton particles (Lee et al., 2013). The presence of 20-$\mu$m nylon fragments in *Calanus helgolandicus* fecal pellets altered their sinking rate but was dependent on fecal pellet volume and type of plastic ingested (Coppock et al., 2019). Fecal pellets containing low density polyethylene particles (10-20 $\mu$m) sank significantly slower than controls, where pellets containing high density polyethylene terephthalate particles sank significantly faster (Coppock et al., 2019). These results are similar to a study by Cole et al. (2016), where polystyrene microplastics encased in fecal pellets reduced fecal density and sinking velocity. Fecal pellets containing polystyrene microplastics had a lower average density of 1.13 +/- 0.01 g $cm^{-3}$, and lower sinking velocities of 38.3 +/- 2.6 m $day^{-1}$ compared to the control; 1.26 +/- 0.01 g $cm^{-3}$ density, and a sinking rate of 86.4 +/- 4.0 m $day^{-1}$ (Cole et al., 2016). There was no significant difference in the size of the fecal pellets, however fecal pellets containing microplastics became fragmented during the experiment (Cole et al., 2016). Fragmentation of fecal pellets may result in the release of microplastics back into the euphotic zone where zooplankton are abundant; this can allow for the re-uptake of microplastics and incorporation in more fecal pellets, thus further studies are required to understand these detriments to the biological pump.

### 1.6. Microplastics in the Marine Food Web

With microplastics polluting 88% of pelagic surface waters (Cozar et al., 2014), these particles are highly bioavailable to marine organisms, either through direct ingestion as discussed previously, or indirectly by trophic transfer through the consumption of contaminated prey (Nelms et al., 2018). Contamination of marine food webs by microplastics are facilitated by the location of oceanic zooplankton, bioavailability of plastic particles, microplastic polymer characteristics, and the feeding strategies and mechanisms of marine biota (Setälä et al., 2018). The trophic transfer of plastic particles is an issue of concern, but the capacity for microplastics to absorb pollutants from surrounding waters, as well as leach chemicals, is also a serious threat that can be biomagnified in a food web (Sharma & Chatterjee, 2017). Even for a short time, if plastics are retained inside an organism or if harmful chemicals are leaking from the particles, bioaccumulation can take place if microplastics are ingested (Setälä et al., 2018). Concerns about the transfer of microplastics between trophic levels have resulted in laboratory studies being carried out to demonstrate the impacts of microplastics on marine biota (Auta et al., 2017); however, the full extent of plastic bioaccumulation in marine biota remains to be known.

As previously stated, it has been well established that microplastics have been ingested at the base of the food chain in a large variety of planktonic organisms such as zooplankton (reviewed by Botterell et al., 2019; Cole et al., 2011, 2013, 2016, 2019), larval fish (Foley et al., 2018), salps (Moore et al., 2001), and amphipod crustaceans (Cau et al., 2019; Iannilli et al., 2018; Jamieson et al., 2019; Ugolini et al., 2013). Many lower trophic organisms like zooplankton filter feed, explaining how these particles can enter the food chain at this level. Microplastics have been found to inhibit growth, chlorophyll production, and photosynthesis of microalgae (Prata et al., 2019), which can decrease energy available for primary consumers. Conversely, aged microplastic particles have been found to procure a biofilm, which increased zooplankton consumption (Lobelle & Cunliffe, 2011; Vroom et al., 2017). This increases the likelihood of microplastic consumption, and decreases growth of natural prey, leading to a rise in the amount of microplastics at the lowest trophic level.

Microplastics have also been discovered at higher trophic level organisms such as oysters (*Ostrea edulis*) (Green, 2016), green sea turtles (*Chelonia mydas*) (Bugoni et al., 2001), seabirds (Tourinho et al., 2010), and fish such as Japanese medaka (*Oryzais latipes*) (Rochman et al., 2014) and the Whitemouth Croaker (*Micropogonias furnieri*) (Arias et al., 2019). There is some speculation as to how these microplastics were ingested by higher trophic organisms; larger marine organisms may not filter feed, so they must consume organisms contaminated with microplastics and/or mistake plastic particles as prey. After consumption, plastics can persist inside an organism for an extended period, making consumption vectors sometimes difficult to ascertain. In the blue mussel, *Mytilus edilis*, plastic particles were translocated from the gut to the circulatory system, where they persisted for over 48 days (Browne et al., 2008). This long retention period increases the likelihood of consumption of these contaminated organisms by predators, heightening biomagnification in oceanic food webs.

Studying trophic transfer of microplastics to higher trophic levels has been difficult due to ethical constraints of subjecting vertebrate animals to laboratory experimentation (Nelms et al., 2018). It is still important to study trophic accumulation of microplastics, so studies have found ways to evade harm to animals, for example, studying scat. Microplastics were transferred by consumption of wild caught Atlantic mackerel (*Scomber scombrus*) to grey seals (*Halichoerus grypus*), and the presence of microplastic particles in seal scats was attributed to the occurrence of trophic transfer from prey to a marine top predator (Nelms et al., 2018). Approximately half of the scat samples contained 1-4 microplastic particles with an average size of 1.5 mm, with ethylene propylene being the most frequent plastic polymer (Nelms et al., 2018). Whether the particles were directly consumed by the fish or underwent trophic transfer from ingestion of contaminated zooplankton is not known in this study; however, either scenario proves transfer of plastics between trophic levels.

Trophic transfer has also been studied in marine invertebrates and vertebrates. An artificial food chain was established starting with *Artemia* sp. (brine shrimp) nauplii to study the transfer of microplastic particles and persistent organic pollutants (POPs) to zebrafish (*Danio rerio*) (Batel et al., 2016). Microplastic particles (1-20 $\mu$m) accumulated inside *Artemia* sp. nauplii and were transferred to the zebrafish via consumption, and fluorescent tracking of benzo[a]pyrene indicated that POPs associated with microplastics may desorb into the zebrafish intestines (Batel et al., 2016). Similarly, blue muscles (*Mytilus edulis*) were exposed to .5-$\mu$m fluorescent polystyrene microplastics and fed to green shore crabs (*Carcinus maenas*) (Farrell & Nelson, 2013). Microplastics were present in the hepatopancreas, ovaries, stomach, and gills of the crabs after only one hour of contaminated blue muscle exposure and persisted in the crabs for almost 21 days thereafter (Farrell & Nelson, 2013). Microplastics persisting inside an organism for an extended period can increase the probability of biomagnification of plastics in the marine food web (Magnusson et al., 2016). Microplastics were detected in the gut of wild-caught Norway lobster (*Nephrops norvegicus*), as well as lobsters in a laboratory setting, after being fed fish contaminated with plastics (Murray & Cowie, 2011). Eighty three percent of the wild-caught lobsters had plastic in their stomachs, and after 24 hours, 100% of the animals fed plastic-seeded fish contained plastics in their stomachs (Murray & Cowie, 2011). Some of the filaments were unable to pass through the lobster's gastric mill system (stomach) and were identified as being plastic polymers used in the fishing industry as ropes or nets (Murray & Cowie, 2011). The authors believe that the likely route for plastic ingested by the lobsters is via accidental ingestion of benthic microplastics during feeding or by consuming contaminated prey (Murray & Cowie, 2011).

### 1.7. Zooplankton and the Ocean Carbon Cycle

Within the complex network of the pelagic food web, zooplankton are one of the critical links between atmospheric carbon dioxide ($CO_2$) and the ocean euphotic zone through the transport of carbon into deeper waters (Steinberg & Landry, 2017). The coastal carbon cycle is also an important sink and transport vector for organic and inorganic carbon and has been affected by anthropogenic activities (Bauer et al., 2013); this cycle can be impacted by estuarine copepods such as *Acartia tonsa*. At the surface of the ocean, phytoplankton convert dissolved inorganic carbon (DIC) into particulate organic carbon (POC) via photosynthesis (Cavan et al., 2019), which is then consumed by herbivorous zooplankton (Duckloe et al., 2001). Particulate organic matter (POM) is then exported away from the surface layer into the ocean's interior by a combination of sinking algae or fecal particles produced by zooplankton and/or vertical mixing of dissolved organic matter by upwelling or upward migration of marine organisms (Hansell, 2002; Roman et al., 2014; Turner, 2015). Coupling these processes together creates what is known as the biological pump, which is responsible for long-term sequestration of what was once atmospheric carbon (Passow & Carlson, 2012). This biological process modulates fluxes of earth's climate, and it is unclear how anthropogenically driven climate change, including warming, acidification, and deoxygenation of ocean waters, will affect the efficiency of the biological pump (Honjo et al., 2014). The biological pump occurs over a wide range of spatial and temporal scales; there are numerous biological, chemical, and physical processes involved, forming a complex system, influencing the earth on a global scale (Honjo et al., 2014).

The ocean contains about 40,000 billion tons of carbon, about 50 times the size of the atmospheric reservoir, which is fed considerably by Anthropogenic carbon (Bopp et al., 2020). Oceanic $CO_2$ trends for 1981-2007 were modeled, and it was estimated that climate change and temperature variability reduced oceanic $CO_2$ uptake by 12%; this reduction is caused by global change, specifically in wind patterns and ocean warming (Le Quéré et al., 2010). With ocean surface temperatures increasing an average rate of .13°C per decade (Laffoley & Baxter, 2016) and high-wind hurricane events occurring more frequently (Knutson et al., 2019), this figure could have underestimated the reduction in $CO_2$ oceanic uptake. More recently, models were created to simulate climate change and its effects on factors such as surface (algal community structure) and subsurface (shift in zooplankton community structure) features; it was found that the downward POC flux was halved, potentially significantly reducing oceanic carbon sequestration (Boyd, 2015). These models did not take into the account the effects of oceanic pollutants such as microplastics, and how these particles affect zooplankton productivity and efficiency.

Microplastics and their effects on the zooplankton community may highly impact the efficiency of the biological pump, where efficiency is noted by Ducklow et al. (2001) as the total carbon transported from the ocean's surface divided by dissolved carbon captured through photosynthesis.

Marine organisms that contribute to the success of the biological pump come in all shapes and sizes and are across many taxa. Large whales can transport nutrients to surface waters by vertical mixing and releasing fecal plumes (Roman et al., 2014; Roman & McCarthy, 2010), and although small, zooplankton perform the same ecosystem services (Paffenhöfer et al., 2001; Steinberg & Landry, 2017; Turner, 2015). Zooplankton heavily contribute to the efficiency of the biological pump through grazing on carbon rich particles, breaking up large marine aggregations, active transfer of POC via diel vertical migration (DVM), and fecal egestion (Cavan et al., 2017; Hansen & Visser, 2016). After modeling carbon transport in the North Atlantic Ocean where spring blooms are prominent, it was found that 27% of the total export flux is transported by migrating zooplankton (Hansen & Visser, 2016). When zooplankton graze on surface phytoplankton and metabolize a portion of this biomass deeper in the ocean after DVM, this migration can release $CO_2$ deep in the water column, speeding up the biological pump (Longhurst et al., 1990). Zooplankton fecal pellets are also an important driver in the biological pump. The composition, size, density, and sinking rate of fecal pellets are affected by diet and can differ in sinking abilities, thus rates of carbon storage (Steinberg & Landry, 2017; Turner, 2002). Fecal pellets are estimated to contribute approximately 40% of total POC, which is a substantial portion of total oceanic carbon flux (Turner, 2015). These findings support that the migration of zooplankton and their effects on carbon transport is substantial and should be incorporated into total biological transport of carbon in the biological pump.

## 1.8. Conclusion

There has been an increasing interest and concern of microplastics in oceanic ecosystems and their complex interactions with marine biota. With the current demand of plastic products, production trends, and ‘single use’ plastic polymers, oceanic plastic debris is projected to increase. Over the past two decades, studies have produced a wide plethora of microplastics research, expanding on the foundational understanding that microplastics are a ubiquitous marine contaminant. However, there is a continuously expanding list of questions that remain to be answered. Establishing accurate oceanic microplastic concentrations, especially particles smaller than the most commonly assayed particle size of 335 $\mu$m, is critical to understanding true bioavailability of microplastic particles to zooplankton species. This information would be helpful in determining nauplii rate of interaction with plastic particles, which could be very detrimental to larval development. Zooplankton have been found to ingest a diverse range of microplastic sizes and polymers, bringing microplastics into the marine food web, thus making the pervasiveness and effects of these particles vast. Upon entering a food web, biomagnification is a serious threat to the health of marine organisms that could consume contaminated prey. Larger organisms at the top of a marine food web that consume contaminated prey such as fish, can pose a serious health risk to the human population.

The efficiency of the biological pump and carbon sequestration is essential in controlling anthropogenically produced $CO_2$, without this system, there would be 50% more carbon stored in the atmospheric reservoir. Understanding how the biological pump may change in response to anthropogenically produced pollution is important in understanding the productivity of the global carbon cycle. As previously discussed, zooplankton fecal pellets have been significantly affected by the presence and ingestion of microplastics. Many laboratory studies looked at the effects of microplastics on fecal shape, sinking rates, and density of various zooplankton, which play a critical role in the biological pump. This information is extremely important in estimating carbon removal rates by oceanic processes, however questions regarding fecal sinking rates remain unanswered. Further lab studies are needed to better understand the effects of microplastics on fecal sinking rates, and how these compare to fecal pellets devoid of plastic particles. There could potentially be a large difference between sinking rates of fecal pellets full of microplastics, and how much less carbon will be sequestered because of this.

# CHAPTER 2: CONSUMPTION OF MICROPLASTICS IMPACTS THE GROWTH AND FECAL PROPERTIES OF THE MARINE COPEPOD, *ACARTIA TONSA*

## 2.1. Introduction

Human activity has put immense stress on global marine ecosystems, with plastic debris as the source for many environmental problems. Since the start of industrial production, an estimated 8 million metric tons of plastic, including 236,000 metric tons of microplastics, make their way to the oceans annually (Jambeck et al., 2015). Plastic constitutes 80 to 85% of marine debris found floating at the surface or embedded in benthic sediment (Auta et al., 2017; Chiba et al., 2018; Jamieson et al., 2019; Law & Thompson, 2014), and of this, less than 5% has been recovered and processed (Sutherland et al., 2010). While large macroplastics (>5 mm) such as packaging materials, grocery bags, and straws (Andrady, 2011; Besseling et al., 2015; Cressey, 2016) can affect large marine biota such as birds (Fry et al., 1987; Provencher et al., 2014; Rios et al., 2007), whales (De Stephanis et al., 2013; Jacobsen et al., 2010; Unger et al., 2016), and economically important fish species (Miranda et al., 2016; Rochman et al., 2015), the impacts of microplastics have only recently begun to be understood, in part due to the challenges of knowing environmental concentrations. Microplastics (<5 mm), however, also add to oceanic debris but are difficult to see, such that standard quantification techniques of plastic debris seriously underestimate the amount of plastics in the ocean (Barnes et al., 2009; Jambeck et al., 2015).

Microplastics are produced from a variety of polymers, some chosen for their buoyancy properties, others for durability (Cole et al., 2011). Included are low-density plastics that float in water such as polyethylene and polystyrene (Auta et al., 2017; SEA, 2012), with polyethylene highly concentrated in sea surface samples (Choy et al., 2019; Erni-Cassola et al., 2019). Primary microplastics are industrially produced for facial cleansers (FDA, 2015; Fendall & Sewell, 2009), tooth polish, medications (DG Environment Report, 2017), and industrial cleaning abrasives (Verschoor et al., 2016). Secondary microplastics fragment from larger marine debris such as plastic bags, fishing nets, and bottles (Andrady, 2017; Cressey, 2016). Multiple components affect the fragmentation of larger plastics into microplastics; these include both time as well as the combination of weathering, UV degradation, and abrasion (Barnes et al., 2009; Browne et al., 2007; Hidalgo-Ruz et al., 2012; Thompson et al., 2004). These factors increase the abundance of marine litter throughout the water column, effectively increasing their availability and/or encounter rate, to marine organisms (Law & Thompson, 2014). Oceanic transport of these particles include Ekman currents (Kubota, 1994; Kubota et al., 2005; Martinez et al., 2009; Onink et al., 2019) stokes drift (Fraser et al., 2018; Onink et al., 2019), seasonal freezing/thawing of drifting Artic Sea ice (Bergmann et al., 2019; Geilfus et al., 2019; Lusher et al., 2015; Peeken et al., 2018) and windy/storm weather conditions (Lattin et al., 2004). The combination of constant fragmentation and circulation of microplastics can increase organismal exposure to these particles. Some plastic polymers are extremely durable, projecting that most polymers produced today will persist for at least decades, centuries, and possibly millennia (Hopewell et al., 2009), increasing encounter rates even more.

Due to their small size and pervasive nature, microplastics have been ingested by marine organisms such as brown shrimp (Devriese et al., 2015), barnacles (Goldstein & Goodwin, 2013), corals (Hall et al., 2015), oysters (Green, 2016), and zooplankton (reviewed by Botterell et al., 2019; Cole et al., 2011, 2013, 2016, 2019; Coppock et al., 2019; Zhang et al., 2019a), yet there are many organisms where the effects of microplastics have not been identified (Nelms et al., 2018).

Occupying a large section of oceanic species diversity (Pomerleau et al., 2015; Reid & Williamson, 2010), zooplankton are distributed ubiquitously in the world's oceans, inhabiting surface waters to the benthic zone (McManus, 2012).

To understand the effects of microplastic consumption on zooplankton, I used then common coastal/estuarine calanoid copepod *Acartia tonsa,* which have short generation times, and can thrive in laboratory conditions. Small marine organisms such as *A. tonsa* will consume microplastics by either false identification of food, or co-ingestion with algae via a non-selective feeding mechanism (Kiørboe, 2011). Through these feeding mechanisms, the consumption of microplastics has caused starvation, low fecundity, and impeded growth (Cole et al., 2013, 2019; Ottvall & Carlsson, 2016). Microplastics and zooplankton are constantly found in equal amounts in surface trawls where feeding occurs, strongly suggesting that microplastics are in such high densities in the upper water column that they are readily available for zooplankton to ingest (Thompson et al., 2004).

Marine zooplankton provide the critical link between primary producers and secondary consumers in the oceanic food web (Cole et al., 2019). Zooplankton consume carbon that was fixed by planktonic phototrophs (Longhurst & Harrison, 1989), which is then transferred to planktivores (Setälä et al., 2018). Consumption of microplastics, however, reduces algae ingestion by zooplankton (Cole et al., 2019; Coppock et al., 2019) and can cause energy shortages, affecting zooplankton early life stage development. Specifically, consumption of microplastics can affect early life stage copepod (nauplii) development (Jeong et al., 2017) and survival (Lee et al., 2013), but body size, as a proxy for development, has not been studied.

Copepod fecal material substantially contributes to the storage of atmospheric carbon dioxide to the deep ocean via the biological pump (Longhurst & Harrison, 1989); a process by which dissolved inorganic carbon is transformed into organic biomass via photosynthesis, then transported to the ocean interior for eventual storage (Cavan et al., 2019; Duckloe et al., 2001; Hansell, 2002; Turner, 2015). The biological pump is vital to slowing down the rate of climate change; it absorbs and removes $CO_2$ from the atmosphere, absorbing about 30% of anthropogenic carbon emissions in the past decade (Bopp et al., 2020). Fecal pellets sink slower when contaminated with microplastics (Cole et al., 2016; Wieczorek et al., 2019), potentially slowing carbon transport. However, modeling the combination smaller fecal pellets and slower sinking rates and their effects on carbon storage rates has yet to be done. Here, I test the hypothesis that the consumption of polystyrene microplastic (5 $\mu$m) particles affect the growth of the copepod *A. tonsa* during the nauplii stage through reduced body size. Additionally, I test the hypothesis that contaminated fecal pellets are smaller in size and sink slower, thus lengthening the time it takes for fecal pellets to settle, thus affecting carbon storage. I extend our results to a theoretical experiment where I modeled the effects of microplastics to the biological pump.

## 2.2. Materials and Methods

### 2.2.1. Copepod Sampling

Zooplankton were sampled from Cedar Beach, Long Island (40°57'51.6" N 73°02'33.4" W) and Peirce Island, New Hampshire (43°4’30.99’’ N, 70°44’58.03” W) in the months of June and July 2018, respectively. Individuals were collected with a horizontally submerged 200-$\mu$m mesh plankton net. The samples were then transferred to 5-L buckets inside a cooler which was insulated in ice and transported to the University of Vermont (UVM) within the same day. Upon arrival, adult *Acartia tonsa* were manually sorted with a 3-mL transfer pipette from the rest of the zooplankton species under a Leica M80 inverted light microscope. After sorting, *A. tonsa* were transferred to a 3-L container of aerated 30 ppt sea water (ASW) made with Instant Ocean (Instant Ocean Spectrum Brands 3001 Commerce St. Blacksburg, VA 24060-6671) salt. Individuals were stored in a temperature-controlled laboratory for a minimum of three generations, in a 12:12 light dark cycle, at 18°C.

### 2.2.2. Algal Cultures

Three algal species were selected for experimental use; they are natural prey for *A. tonsa* and have a similar size to the microplastic particles used in this study. Chain-forming diatom *Thalassiosira weissflogii* (2-32 $\mu$m), red microalgae *Rhodomonas lens* (max 12 $\mu$m), and chlorophyte *Tetraselmis* sp. (5-14 $\mu$m) were used. All algal species were cultured at UVM using F/2 media (Kent Marine, Pro-Culture), with additional silicates (sodium metasilicate nonahydrate, 30 g $L^{-1}$ in reverse osmosis water) for *T. weissflogii* under a 10:14 light cycle at ambient temperature. For experiments, algal cells were counted on a Sedgewick Rafter cell slide using a Nikon SMZ-800 dissecting scope for particle concentrations, and 167 $\mu$g C $L^{-1}$ of each species was combined to add 501 $\mu$g C $L^{-1}$ to each experimental replicate. Microplastic particle concentrations were calculated to equal a 1:1 ratio of microplastics to the number of algae particles. This resulted in double the particle density in the treatment, but the same density of algae particles in both the control and the treatment replicates.

### 2.2.3. Microplastics

SPHERO red polystyrene particles were purchased from Spherotech Inc, in the size range of 6.0-8.0 $\mu$m with a mean particle size of 6.69 $\mu$m. This particle range encompasses all three algal species size ranges. Before using, particles were resuspended by vortexing and vigorous shaking. The particle suspension matrix also included 0.02% sodium azide. To ensure that the suspension matrix alone did not affect survival of *A. tonsa*, I measured survival with and without the microplastic suspension mixture and found that survival was unaffected after multiple trials.

### 2.2.4. Fecundity Analysis

Treatments comprised of a control (strictly algal exposure), and a 1:1 ratio of algal particles to microplastic particles. One hundred thirty-six copepodite *A. tonsa* (juvenile stage CV) were sorted and incubated in 1-L Tupperware beakers filled with 600 mL of ASW (4 beakers per treatment, 17 individuals in each). Exposures to microplastics occurred for 7 days at 18°C in a 12:12 light/dark cycle, when the copepodites reached adulthood. Water changes were done every 72 hours where 50% of media was removed by pouring beaker contents over a 30-$\mu$m mesh to retain copepodites. Afterwards, fresh ASW, 501 $\mu$g C $L^{-1}$ of algae, and corresponding microplastic particle concentrations were added to the treatment replicates. I acknowledge in that only removing 50% of media per water change, microplastic and algae particle densities may have slightly increased over time, however this increase occurred in all replicates, and the 1:1 ratio of algae and microplastic particles was conserved. Post exposure, adults were carefully poured over a 30-$\mu$m sieve, assayed for survival, and sexed under a Leica M80 inverted light microscope. A single adult breeding pair was placed in a petri dish with 25 mL of ASW, conserving treatments and replicates. Five hundred one $\mu$g C $L^{-1}$ of combined algae was added to all petri dishes, and corresponding microplastic particles were added to petri dishes containing adults that were exposed to microplastics as copepodites. Petri dishes were incubated at 18°C in a 12:12 light/dark cycle for 48 hours to allow adults to lay eggs. Post exposure, contents of the petri dishes were anaesthetized and stained using Lugol's Iodine, then imaged, counted, and measured using Lecia SPOT Measuring Software under an Olympus iX71 inverted light microscope.

### 2.2.5. Body Size Analysis

Treatments comprised a control (strictly algal exposure), and a 1:1 ratio of algal particles to microplastic particles. Forty *A. tonsa* nauplii were introduced to each Tupperware beaker (3 beakers per treatment) containing 400 mL of ASW. Exposures to microplastics occurred for 5 days at SST in a 12:12 light/dark cycle. Fifty percent water changes were done every 48 hours by reverse syphoning beaker media through a 30-$\mu$m sieve. Afterwards, fresh ASW, 501 $\mu$g C $L^{-1}$ of algae, and corresponding microplastic particle concentrations were added to the treatment replicates. Post exposure, the contents of each beaker were poured over a 30-$\mu$m sieve to collect nauplii, which were then carefully sprayed down into petri dishes and assayed for survival. Contents of the petri dishes were anaesthetized and stained using Lugol's Iodine. Five nauplii from each replicate (15 control and 15 plastics) were imaged and measured using Lecia SPOT Measuring Software under an Olympus iX71 inverted light microscope.

### 2.2.6. Fecal Analysis

As per the previous experimental setups, treatments comprised of a control (strictly algal exposure), and a 1:1 ratio of algal particles to microplastic particles. Two hundred *A. tonsa* eggs were introduced to each 1-L Tupperware beaker (4 beakers per treatment) containing 600 mL of ASW. Fifty percent water changes were done every 72 hours by reverse syphoning beaker contents through a 30-µm sieve. Afterwards, fresh ASW, 501 $\mu$g C $L^{-1}$ of algae, and corresponding microplastic particle concentrations were added to the treatment replicates. Copepods were incubated until the final molt to the adult stage (20 days) at SST in a 12:12 light/dark cycle. Adults were placed in groups of 2 or 3 in petri dishes, 1:1 ratio of algae and microplastic particles were added per 25 mL of ASW and let alone for 48 hours. Post exposure, all fecal pellets in each petri dish were counted, imaged, and measured using Lecia SPOT Measuring Software under an Olympus iX71 inverted light microscope. Fecal volume was calculated using the equation for volume of a cylinder, $V = \pi r^2 h$, where the radius was half the width. To estimate the reduction in sinking rates of *A. tonsa* fecal pellets contaminated with microplastics compared to pellets strictly containing natural algae, I used a modified Stokes equation (Komar et al., 1981) for sinking cylinders at low Reynolds numbers

$$\omega_s = 0.0790 \frac{1}{\mu} (\rho_s - \rho) g L^2 (\frac{L}{D})^{-1.664}$$

where $\mu$ is the viscosity of 30 ppt sea water at 18°C (g $cm^{-1}$ $s^{-1}$), $\rho s$ is fecal pellet density (g $cm^{-3}$), $\rho$ is the density of 30 ppt sea water at 18°C (g $cm^{-3}$), $g$ is the acceleration due to gravity (981 cm $s^{-2}$), and L is the length and D is the diameter of fecal pellets (cm) from my experiments. I used the pellet density as reported by Cole et al. (2016) (control: 1.26 +/- .01 g $cm^{-3}$, plastic: 1.13 +/- .01 g $cm^{-3}$), who used the same plastic polymer as in this study, polystyrene, which has a density of ~1.05 g $cm^{-3}$ (Cole et al., 2016). I acknowledge that Cole et al. (2016) used a different species (*Calanus helgolandicus*) than my study organism (*Acartia tonsa*), however both species are coastal calanoid copepods (Prog, 1983; Bonnet, 2005).

#### 2.2.7. Statistical Analysis

All data analysis was conducted using R version 3.6.3. To analyze fecal, fecundity, and body size data, the *lme* function from R package *nlme* was utilized to perform a linear mixed-effects analysis to compare the relationship between the two treatments, modeling replicate as a random effect. *P*-values were obtained by running a Type II Wald Chi Square test. Using function *glmer* from the R package *lme4*, a generalized linear mixed-effects model was used for survival analysis. Utilizing a binomial distribution, survival was entered as either 'alive' (1) or 'dead' (0), with replicate modeled as a random effect. Significant differences were confirmed where $P < 0.05$.

## 2.3. Results

### 2.3.1. Microplastic Uptake

Both nauplii and adult marine copepod *A. tonsa* ingested polystyrene microplastics. Microplastics were observed in adult intestinal tracts and egested fecal pellets, confirming consumption and successful egestion of the polystyrene beads (Figure 1A-C), which is consistent with the findings of previous studies (Cole et al., 2013; Cole et al., 2016; Coppock et al., 2019; Lee et al., 2013). Microplastics were also ingested by nauplii (Figure 1D) and adhered to adult *A. tonsa* swimming legs (Figure 1E).

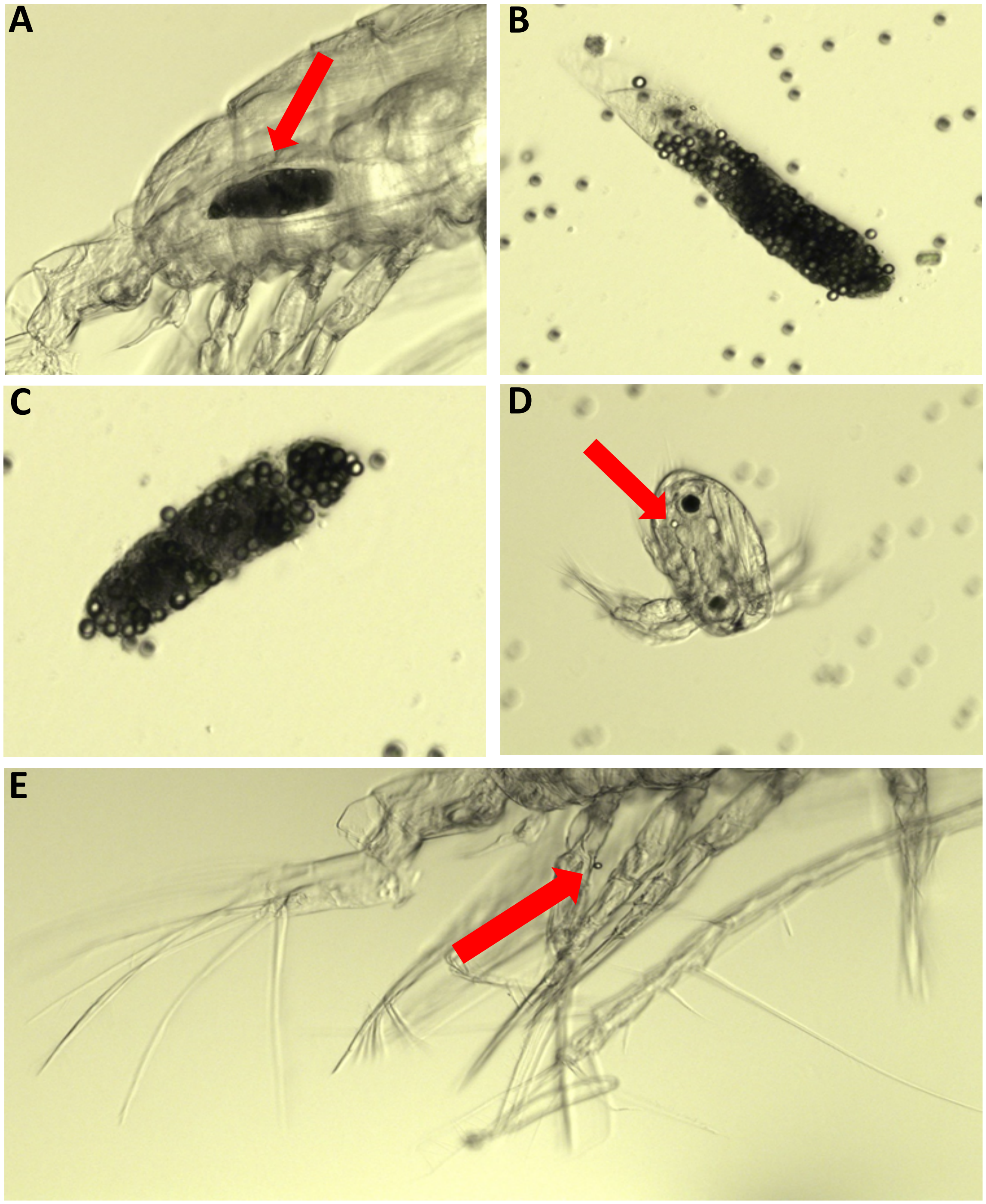


Figure 1: (A) Fecal pellet laden with 5-µm polystyrene microplastics inside the hindgut of copepod *Acartia tonsa*; (B) Fecal pellet containing microplastics egested by adult *A. tonsa*; (C) fragmented microplastic fecal pellet egested by adult copepod. (D) *A. tonsa* nauplii with microplastic bead in its hindgut; (E) microplastic trapped in hind swimmers of adult. All microplastic concentrations were a 1:1 ratio of algae to 5-$\mu$m microplastic particles.

### 2.3.2. Fecundity

After rearing copepodites (juvenile *A. tonsa*) to the adult stage in the presence of polystyrene microbeads, eggs produced by adults had a 7.3% reduction in diameter (control: 79.07 +/- 3.27 $\mu$m; plastic: 73.32 +/- 6.32 $\mu$m; Figure 2A; Linear mixed-effects model, df = (4, 80), $\chi^2$ = 30.03, t-value = -5.48, $P < 0.001$). No effect on copepodite survival to the adult stage was observed (Figure 2B; Generalized linear model, df = 100, z = -0.50, $P = 0.28$). In addition, the presence of microplastics did not affect total fecundity in the measured 48-hour period (Figure 2C; Linear mixed-effects model, df = (4, 12), $\chi^2$ = 1.30, t-value = -1.14, $P = 0.32$).

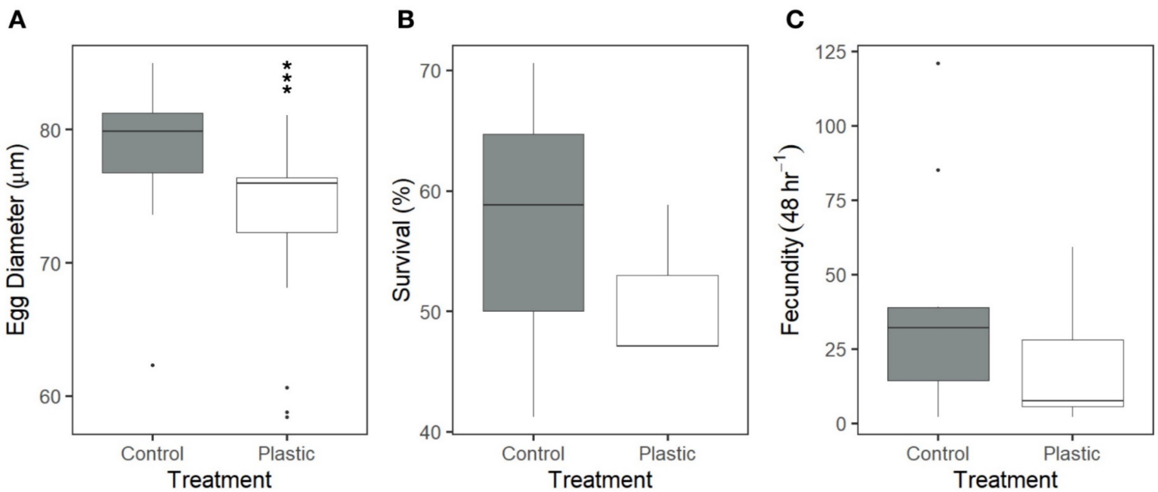


Figure 2: The impacts of microplastics on reproduction characteristics of *Acartia tonsa* adults. (A) Egg diameters produced by adults in 48 hours (Linear-mixed effects model, df = (4, 80), $\chi^2$ = 30.03, t-value = -5.48, $P < 0.001$); (B) percent survival (Generalized linear model, df = 100, z = -0.50, $P$ = 0.28); (C) total reproductive output after 48 hours (Linear mixed-effects model, df = (4, 12), $\chi^2$ = 1.30, t-value = -1.14, $P = 0.32$) after rearing copepodites (juvenile *A. tonsa*) to the adult stage in the presence of polystyrene microbeads. Treatments: control (grey) and plastic (white); asterisks indicate statistical significance (*$P < 0.05$; ** $P < 0.01$; *** $P < 0.001$).

### 2.3.3. Nauplii

After a 5-day exposure period, in the presence of microplastics, *A. tonsa* nauplii had a 12.2% decrease in body length compared to the control (control: 151.28 +/- 15.20 µm; plastic:

172.28 +/- 29.32 $\mu$m; Figure 3A; Linear mixed-effects model, df = (4, 24), $\chi^2 = 4.98$, t-value = -2.23, $P < 0.05$), while no effect on body width in individuals exposed to microplastics was observed (76.19 +/- 8.32 $\mu$m) compared to the control (82.12 +/- 15.29 $\mu$m); (Figure 3B; Linear mixed-effects model, df = (4, 24), $\chi^2 = 1.03$, t-value = -1.01, $P = 0.31$). In contrast, an effect of microplastics exposure on survival was detected (Figure 3C; Generalized linear model, df = 239, z = -2.75, $P < 0.01$).

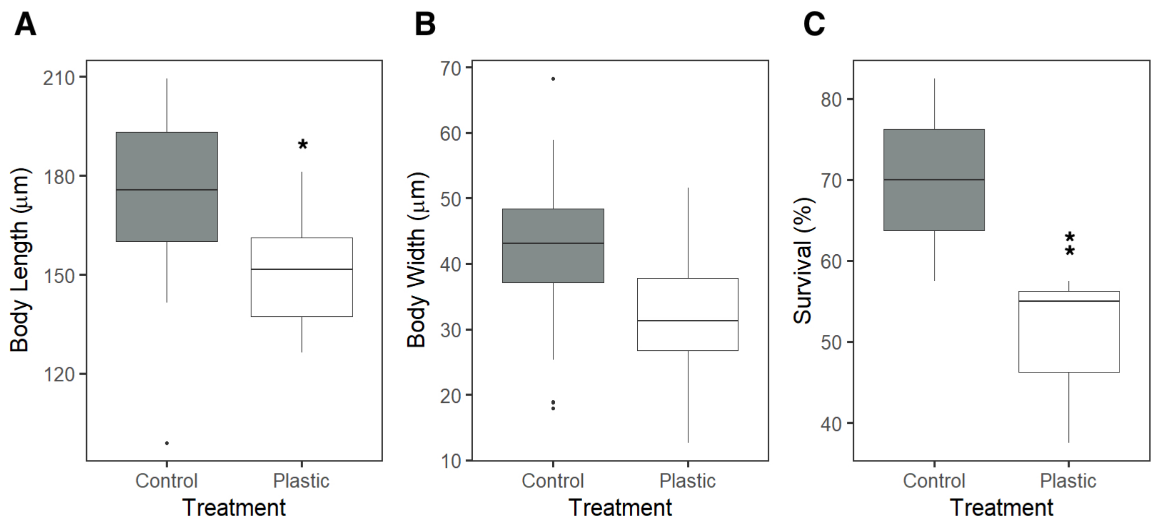


Figure 3: The impact of microplastics on the body size of *Acartia tonsa* nauplii. (A) Nauplii *A. tonsa* body length (Linear mixed-effects model, df = (4, 24), $\chi^2 = 4.98$, t-value = -2.23, $P < 0.05$); (B) width (Linear mixed-effects model, df = (4, 24), $\chi^2 = 1.03$, t-value = -1.01, $P = 0.31$); (C) percent survival (Generalized linear model, df = 239, z = -2.75, $P < 0.01$) after a 5-day exposure to polystyrene microbeads. Treatments: control (grey) and plastic (white); asterisks indicate statistical significance (*$P < 0.05$; ** $P < 0.01$; *** $P < 0.001$).

### 2.3.4. Egested Fecal Pellets

Incorporation of 5-µm polystyrene microplastics resulted in a 26.8% decrease in fecal length (control: 158.19 +/- 38.04 $\mu$m; plastic: 115.74 +/- 36.02 $\mu$m; Figure 4A; Linear mixed-effects model, df = (5, 353), $\chi^2 = 10.72$, t-value = -3.27, $P < 0.001$), and a 24.7% reduction in width (control: 42.62 +/- 8.24 $\mu$m; plastic: 32.10 +/- 7.48 $\mu$m; Figure 4B: Linear mixed-effects model, df = (5, 353), $\chi^2$ = 9.92, t-value = -3.15, $P$ < 0.01) of fecal pellets egested by adults that had developed in the presence of microplastics since the egg stage. In the absence of microplastics, adult fecal pellets had an average volume of 247948.8 +/- 133123.7 $\mu m^3$. Fecal pellets containing microplastics had volumes reduced by 56.4% averaging 108071.7 +/- 77937.5 $\mu m^3$ (Figure 4C; Linear mixed-effects model, df = (5, 353), $\chi^2$ = 12.50, t-value = -3.54, $P$ < 0.001). Although not quantified, a noticeable amount of fragmented fecal pellet pieces that could have broken off during the process of egestion was observed (Figure 1C).

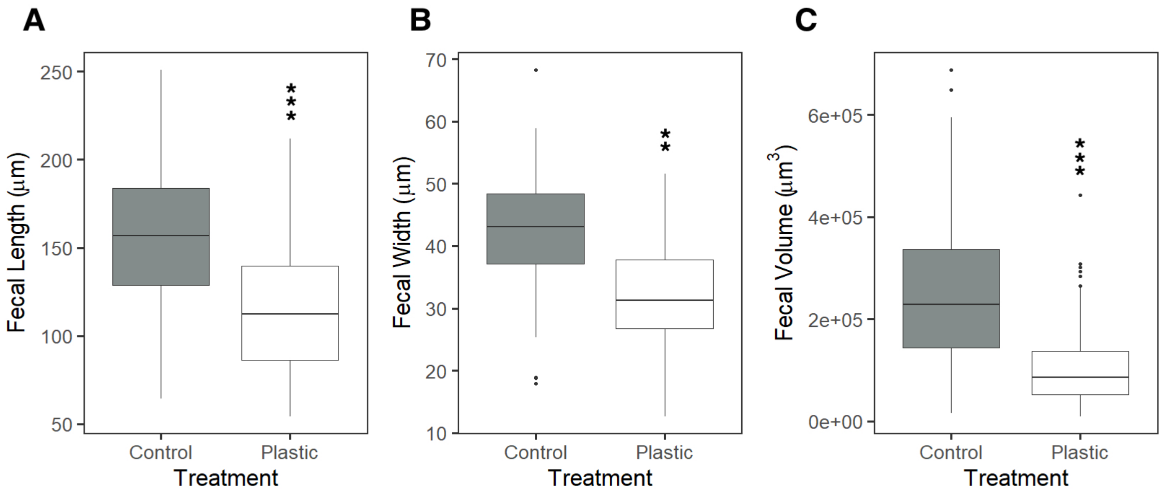


Figure 4: Impacts of microplastic consumption on egested fecal pellet length, width, and volume produced by adult *Acartia tonsa*. (A) Fecal pellet length (Linear mixed-effects model, df = (5, 353), $\chi^2$ = 10.72, t-value = -3.27, $P$ < 0.001), (B) fecal pellet width (Linear mixed-effects model, df = (5, 353), $\chi^2$ = 9.92, t-value = -3.15, $P$ < 0.01), (C) total fecal pellet volume (Linear mixed-effects model, df = (5, 353), $\chi^2$ = 12.50, t-value = -3.54, $P$ < 0.001). Treatments: control (grey) and plastic (white); asterisks indicate statistical significance (*$P$ < 0.05; ** $P$ < 0.01; *** $P$ < 0.001).

### 2.3.5. Sinking Rates

After using the modified stokes equation, contaminated fecal pellets were calculated to sink 3.73 times slower than pellets containing natural algae (control: 41.77 +/- 15.21 m day$^{-1}$; plastic: 10.83 +/- 4.98 m day$^{-1}$; Figure 5; Linear mixed-effects model, df = (5, 353), $\chi^2$ = 75.15, t-value = -8.67, $P < 0.001$).

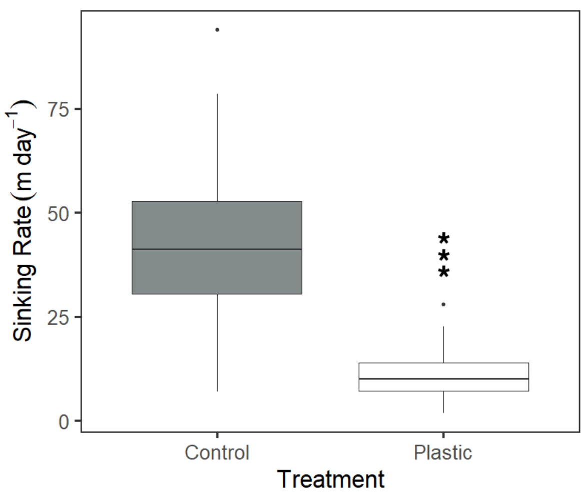


Figure 5: The effect of microplastic consumption on fecal pellet sinking rates of *A. tonsa* (Linear mixed-effects model, df = (5, 353), $\chi^2$ = 75.15, t-value = -8.67, $P < 0.001$) Treatments: control (grey) and plastic (white); asterisks indicate statistical significance (*$P < 0.05$; ** $P < 0.01$; *** $P < 0.001$).

## 2.4. Discussion

My results suggest, for the first time, that microplastics can significantly affect the rate of carbon storage due to shorter, smaller fecal pellets that sink slower. My data also demonstrate that microplastic consumption can hinder the growth of *A.tonsa* by a reduction in body length and reduction in survival of nauplii copepods, highlighting that marine microplastics can affect multiple life history stages. Polystyrene microplastics (5 $\mu$m) were consumed by both nauplii and adult stage copepods and were observed in both the anterior and posterior portions of their intestinal tracts (Figure 1A, Figure 1D). These particles were also found trapped in the external appendages of live copepods (Figure 1E), as well as encapsulated in both fragmented and whole fecal pellets (Figure 1B, Figure 1C). *A. tonsa* adults that consumed microplastics produced eggs with smaller diameters.

### 2.4.1. Fecundity

Consuming algae rich with carbon and nitrogen is essential for copepod growth and egg production (Kuijper et al., 2004). After consumption, up to 85% of algae carbon biomass is used for growth, specifically egg production, in adult females of *A. tonsa* (Kiørboe, 2008). Additionally, 55% of egg biomass comprises proteins, where nitrogen is essential for protein synthesis (Kiørboe et al., 1985). Such composition highlights the importance of carbon and nitrogen consumption for viable reproduction. The quality (i.e. C:N ratios) and quantity of algae biomass ingested and egg production in marine copepods are directly related, which when disrupted, can cause decreased fecundity and smaller eggs (Nobili et al., 2013).

A 7.3% reduction in the diameter of eggs produced as a result of consuming microplastics could be attributed to less carbon biomass consumed because *A. tonsa* misidentified microplastics for algae particles or random co-ingestion of microplastics and algae. Visually less biomaterial was observed in contaminated fecal pellets egested by adult *A. tonsa*, which supports the hypothesis that the consumption of microplastics decreased carbon and nitrogen available for egg production. Smaller eggs could result in stunted growth, high naupliar mortality before the copepodite stage, or could permanently decrease the individual's potential size at adulthood.

My results of smaller eggs as a result of microplastic consumption are comparable to observed reductions in egg diameters of *Calanus helgolandicus* in the presence of microplastics

(Cole et al., 2015). However, microplastic exposure for *C. helgolandicus* started at the onset of the adult stage (Cole et al., 2015), which differs from my study in that exposures started in early copepodite stages - the stage when females begin egg production (Norrbin, 1994). The first traces of female gonad oogenesis are during the first of six copepodite stages of *Acartia* sp. (Norrbin, 1994). Oocytes continue to grow in the developing copepodites to the adult stage (Mauchline, 1998) where the female's gonads develop in the presence of microplastics. This could explain why my study observed a more significant effect on egg diameter than Cole et al. (2015). To better understand the relationship between egg size, nutrition, and microplastic consumption, I suggest full life exposures to microplastics to assay body length and lipid content at each stage.

I found no difference in fecundity (total eggs produced), hatching rates, or survival in the experimental time frame of 48 hours, which concurred with the results from Bellas & Gil (2020) after a 48-hour exposure period to polyethylene microplastics to *A. tonsa*. Along with Bellas & Gil (2020), I suggest that a more chronic exposure period of *A. tonsa* adults to microplastics may reveal significant effects on fecundity, hatching rates, and survival, which should be tested using the same microplastic concentrations for future experiments.

### 2.4.2. Nauplii

As previously stated, microplastics and zooplankton are consistently found in equal amounts in surface trawls where feeding occurs, suggesting that microplastics are readily available for zooplankton to ingest (Thompson et al., 2004). Common microplastic quantification techniques use surface collection nets too large to assay particles smaller than 335 $\mu$m (Thompson et al., 2004), missing microplastics in mid-water and benthic zones, thus underestimating true oceanic microplastic concentrations (Andrady, 2011). Newly hatched *A. tonsa* nauplii are approximately

70 $\mu$m in length (Marcus & Wilcox, 2007), falling in the size range of infrequently environmentally assayed microplastics.

The true impacts of microplastic consumption on this vulnerable stage is unknown. Consumption of microplastics affect copepod early life stage (naupliar) development through delayed molting (Jeong et al., 2017) and reduced survival (Lee et al., 2013), which differs from the findings reported here. As previously discussed, a reduction in development could be from less carbon biomass consumed due to the consumption of microplastics instead of algae particles, or the co-ingestion of microplastics and algae. Maximum algal particle clearance rates for nauplii stages II-IV has been determined to be approximately 7 $\mu$m, (Berggreen et al., 1988), suggesting that nauplii actively select and consume microplastic particles 7 $\mu$m and smaller. This heightens the possibility that nauplii are consuming large numbers of small, under assayed microplastic particles.

The molt from the final naupliar stage (N6) to the first copepodite stage (C1) is perhaps the most significant in the life cycle of *A. tonsa* (Leandro et al., 2006), as well as other zooplankton species. After exposure to 0.5 $\mu$m microbeads, the copepod *Paracyclopina nana* had delayed molting from the naupliar to copepodite stage (Jeong et al., 2017). Reduced growth rates could prolong the amount of time an induvial exists in the naupliar stage, heightening the possibility of delayed or failed molting, or size-selective predation. Nauplii are subject to high predation rates due to their weak ability to detect predators and underdeveloped swimming appendages, yielding a prolonged escape response (Sell et al., 2001). This could lower the proportion of nauplii that survive to the copepodite stage, and in turn reduce the number of reproducing adults, possibly affecting population growth. Additionally, nauplii that have reduced survival and energetic shortages (reduced lipid content) due to microplastic exposure may also impact higher trophic organisms that prey on early stage zooplankton (Cole et al., 2015). This could cause an energy imbalance at the start of the oceanic food chain, potentially cascading energy shortages into further trophic levels.

### 2.4.3. Fecal Properties

To my knowledge, these results clearly demonstrate for the first time that microplastics could significantly alter the rate of carbon storage by reducing fecal volume and fecal sinking rates. These findings are similar to Cole et al. (2016), where fecal pellets egested by *Calanus helgolandicus* had lower densities and slower sinking rates, however there was no difference in fecal size or volume. This critical difference reveals that in the presence of microplastics, *A. tonsa* are consuming less biomass leading to smaller, slower sinking fecal pellets, which can have severe implications on the efficiency of the biological pump, as well as increasing the reuptake of contaminated fecal pellets in the oceanic food web.

Sinking zooplankton fecal pellets heavily contribute to the efficiency of the biological pump, transporting carbon and particulate organic matter (POM) to the ocean's interior and benthic sediment (Cavan et al., 2017; Steinberg & Landry, 2017; Turner, 2002). Zooplankton fecal pellets are estimated to contribute, on average, approximately 40% of total particulate organic carbon (POC), thus substantially supporting total oceanic carbon flux (Turner, 2015). The biological pump modulates fluxes of earth's climate (Honjo et al., 2014) and is responsible for long-term storage of anthropogenically produced carbon (Longhurst & Harrison, 1988). Our results confirm the hypothesis that microplastics can affect the efficiency of the biological pump, which when coupled with the effects of climate change (i.e. rising sea temperatures and changing wind patterns) (Le Quéré et a., 2010), could drastically reduce ocean carbon storage.

Understanding that *A. tonsa* is a coastal/estuarine species, their fecal pellets were used as a proxy for the effects of microplastics on the biological pump. In extrapolating my data, I modeled the combined effects of smaller fecal volumes and slower fecal sinking rates on carbon storage. The model estimates that 8.85 times more fecal volume settles per day from fecal pellets strictly containing algae (**control**: 3,682 m (average depth of the ocean; Charette & Smith, 2010) / 41.77 m $day^{-1}$ (average fecal sinking rate) = 88.15 days (time for fecal pellets to sink to the benthos); 247,949 $\mu m^3$ (average fecal volume) / 88.15 days = 2,812.81 $\mu m^3$ $day^{-1}$ (rate of total fecal volume sinking per day); **plastic**: 3,682 m (average depth of the ocean; Charette & Smith, 2010) / 10.83 m $day^{-1}$ (average fecal sinking rate) = 339.98 days (time for fecal pellets to sink to the benthos); 108,072 $\mu m^3$ (average contaminated fecal volume) / 339.98 days = 317.88 $\mu m^3$ $day^{-1}$ (rate of total fecal volume sinking per day); 2,812.81 $\mu m^3$ $day^{-1}$ / 317.88 $\mu m^3$ $day^{-1}$ = **8.85** times more fecal volume per day). This calculation is conservative, in that I didn't incorporate the fact that fecal pellets laden with microplastics tend to fragment easily, and that fecal pellets are 100% organic carbon material. Uncontaminated fecal pellets are larger and more densely packed with carbon which significantly increases the speed of carbon transport to the ocean floor (Figure 4A-C, Figure 5, Figure 6C, 6D) increases fecal pellet integrity reducing fragmentation, and realistically contains more carbon than contaminated fecal pellets. In turn, this removes particulate organic carbon from the euphotic zone, reducing the chances that carbon could be released back into the atmosphere as $CO_2$.

A reduction in carbon storage could have huge implications on the size of the atmospheric carbon reservoir, increasing oceanic $CO_2$ abundance. High levels of dissolved $CO_2$ in the ocean has resulted in ocean acidification, affecting marine calcifiers such as sea urchins *Lytechinus variegatus* (Emerson et al., 2017), *Hemi- centrotus pulcherrimus* and *Echinometra mathaei* (Kurihara & Shirayama, 2004). Under elevated oceanic $CO_2$ conditions, spine integrity, fertilization rate, cleavage rate, developmental speed, and pluteus larval size all tended to decrease with increasing CO2 concentration (Emerson et al., 2017; Kurihara & Shirayama, 2004); this greatly affected normal phenotypes, thus adaptations to future environmental stressors may be difficult. Although marine copepods are not calcifiers, elevated levels of $CO_2$ disrupts the phytoplankton community, resulting in smaller and less algal cells available for consumption (Bopp et al., 2005; Morán et al., 2010), affecting zooplankton populations.

Without quantification of feeding selectivity, the differences of fecal pellet size between treatments was still interesting to observe and comparing my results to other studies was still relevant. My 1:1 ratio of microplastic to algae particles is unique for fecal studies and could be a factor in the results discussed in this paper. Microplastic concentrations equal to my experimental algal concentration of 501 $\mu$g C $L^{-1}$ may be higher than average microplastic persistence in the environment, however these concentrations are comparable to other studies (Setälä et al., 2014; Cole et al., 2013; Lee et al., 2013; Vroom et al., 2017). Using a new autofluorescence method for detecting microplastics, a mean plastic concentration of 8277 particles $L^{-1}$ was found in California waters, averaging 5–7 orders of magnitude higher than previous studies (Brandon et al., 2020), proving that with newer technologies, more accurate microplastic concentrations are determined continuously. This validates my justification for using high experimental microplastic concentrations due to the complexities of extraction as well as the vastness of the ocean sampling. It can be deduced that as larger microplastic pieces break down into microplastics, plastic particles exponentially increase, which only heightens uncertainty in quantifying oceanic plastic debris.

Fecal pellets of many marine invertebrates are superficially encased and glued together with mucus to provide a solid pellet shape (Wotton & Malmqvist, 2001). A mucus encasing allows for fecal pellets to remain whole and hold its shape for days, if not weeks (Wotton & Malmqvist, 2001). As found by Cole et al. (2016), I observed an elevated amount of contaminated fecal pellets that were starting to fragment, or upon egestion, broke apart. Slower sinking fecal pellets prolong time spent in the water column, which increases the likely hood for the mucus membrane to break down, releasing microplastic particles, and thus potentially consumed by another organism (Figure 6E).

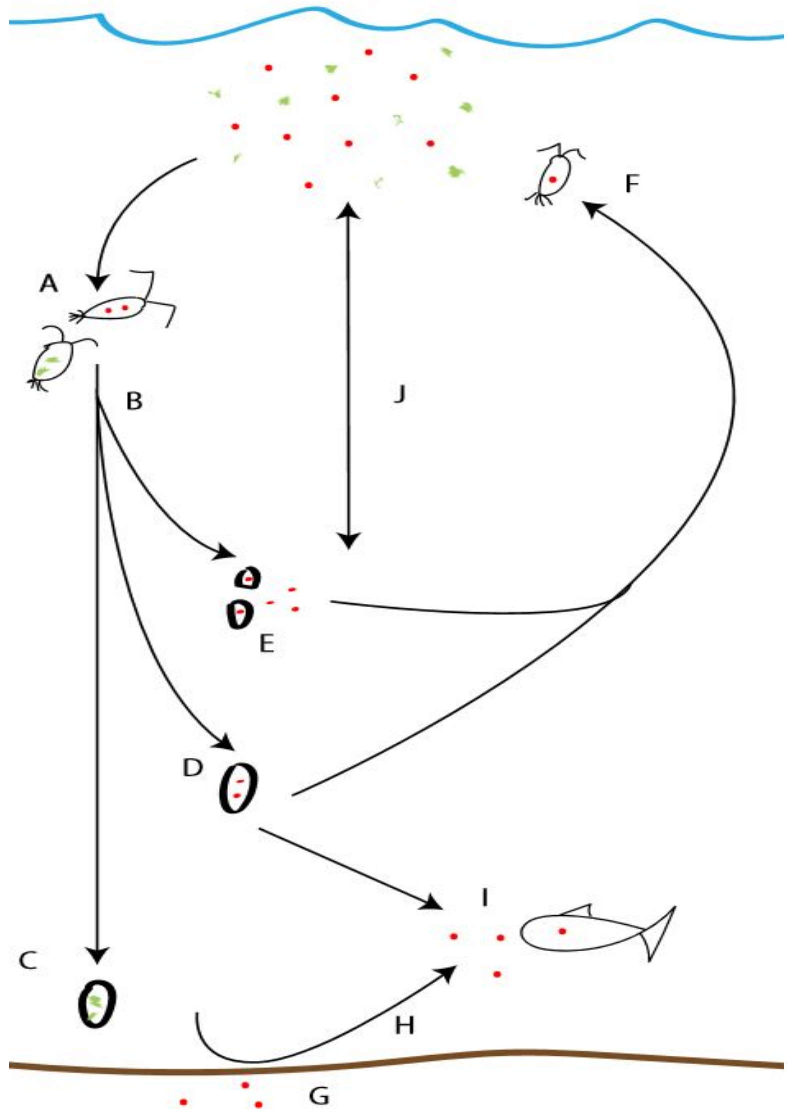


Figure 6: Theoretical representation of low-density microplastic transport vectors via zooplankton in the marine water column. (A) Zooplankton ingest microplastic particles (red dots), either by co-ingestion with algae (green particles) or misidentification of microplastics for prey; (B) zooplankton egest these microplastics in their fecal pellets; (C) fecal pellets containing natural algal prey are more dense, and sink quickly; (D) fecal pellets contaminated with low-density microplastics will sink significantly slower; (E) fecal pellets containing microplastics are more prone to fragmentation due to the lack of dense organic material, releasing microplastic particles into the water column; (F) zooplankton, in diel vertical migration, may ingest free floating microplastics or consume contaminated fecal pellets, thus returning the microplastic particles to the surface; (G) benthic sedimentation of microplastics; (H) microplastics stirred up by upwelling, ocean currents, or scavenging organisms; (I) consumption of microplastics by benthic organisms such as fish; (J) sinking of microplastic particles due to gravity or returned to the surface via oceanic flux.

Coprophagy, or the consumption of feces, is common among aquatic organisms (Frankenberg & Smith, 1967), and can lead to the reuptake of microplastics in the oceanic food web. Sinking slower in the water column, contaminated fecal pellets hold the potential to be consumed at higher rates by coprophagous copepods as well as larger organisms, which also leads to prolonged circulation of microplastics in food webs. Studying how food web dynamics are affected by microplastics is difficult, but one-link trophic transfers has been observed (Batel et al., 2016; Browne et al., 2008; Nelms et al., 2018). Blue muscles (*Mytilus edulis*), a benthic dwelling organism, were exposed to 0.5- $\mu$m fluorescent polystyrene microplastics and fed to green shore crabs (*Carcinus maenas*) (Browne et al., 2008).  Microspheres were present in the hepatopancreas, ovaries, stomach, and gills of the crabs, after only one hour of contaminated blue muscle exposure, and persisted for almost 21 days thereafter (Farrell & Nelson, 2013). Microplastics persisting inside an organism for an extended period might influence the probability of biomagnification of plastics in the marine food web (Magnusson et al., 2016).

### 2.5. Conclusion

Here, I have highlighted that microplastics have the potential to affect the marine environment at the organismal level, and on a global scale with impacting the efficiency of the biological pump and the biomagnification of microplastics in the oceanic food web. My results demonstrate reduced nauplii body length, survival, and egg diameters in the presence of microplastics, stressing the importance to understand the effects of microplastics on all zooplankton life history stages. Smaller egg diameters could lead to smaller adults which may impact reproductive output, thus affecting future generations. With reduced growth rates, nauplii may not reach the adult stage due to slower swim rates, possibly resulting in increased predation, which should be assessed at the organismal level using predation assays. Combined energy losses of reduced growth rates and smaller egg diameters could be amplified to the food web level through a trophic level transfer efficiency analysis, looking at potential energy shortages at higher trophic levels. Decreased survival at the nauplii stage could reduce the number of individuals that reach reproductive maturity, which could impact population growth. Such decreases in population growth should be modeled to better understand the residual effects of low survival at this vulnerable zooplankton stage. This knowledge is vital in understanding how critical zooplankton populations may be affected, as they have been living among oceanic microplastics for some time and are a critical energy link in oceanic food webs.

I also show that fecal pellets contaminated with microplastics have the potential to reduce the efficiency of oceanic carbon storage, which in conjunction with current anthropogenic carbon production, can have major implications for oceans globally. For future models aiming to project the impacts of microplastics on fecal pellets and carbon storage, I recommend that changes in fecal volume and sinking rate be included for realistic estimates. Since microplastics are continuously introduced to the marine environment, zooplankton are predicted to live amongst these particles for multiple generations. In this study, microplastic exposures only occurred during one generation, which may not have captured the full effect microplastics have on fecal pellets. I propose multi-generation exposures may reveal more significant effects on fecal properties, which can reduce carbon settling via sinking fecal pellets more severely than discussed here. A better understanding of microplastic exposure on the structural integrity, vertical distribution, and circulation of fecal pellets is necessary in creating accurately parameterized models to assess the effects of microplastics on the biological pump and carbon settling

## COMPREHENSIVE BIBLIOGRAPHY


Akester, H. (2019). How big a problem is ocean polystyrene pollution? PHYS ORG. https://phys.org/news/2019-12-big-problem-ocean-polystyrene-pollution.html

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

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

Andrady, A. L. (2017). The plastic in microplastics: A review. In Marine Pollution Bulletin (Vol. 119, Issue 1, pp. 12–22). Elsevier Ltd. https://doi.org/10.1016/j.marpolbul.2017.01.082

Archibald, K. M., Siegel, D. A., & Doney, S. C. (2019). Modeling the Impact of Zooplankton Diel Vertical Migration on the Carbon Export Flux of the Biological Pump. Global Biogeochemical Cycles, 33(2), 181–199. https://doi.org/10.1029/2018GB005983

Arias, A. H., Ronda, A. C., Oliva, A. L., & Marcovecchio, J. E. (2019). Evidence of Microplastic Ingestion by Fish from the Bahía Blanca Estuary in Argentina, South America. Bulletin of Environmental Contamination and Toxicology, 102(6), 750–756. https://doi.org/10.1007/s00128-019-02604-2

Ašmonaitė, G., & Almroth, B. C. (2019). Effects of Microplastics on Organisms and Impacts on the Environment: Balancing the Known and Unknown (Issue February). https://doi.org/10.13140/RG.2.2.28556.77448ėš

Auta, H. S., Emenike, C. U., & Fauziah, S. H. (2017). Distribution and importance of microplastics in the marine environment: A review of the sources, fate, effects, and potential solutions. In Environment International (Vol. 102, pp. 165–176). Elsevier Ltd. https://doi.org/10.1016/j.envint.2017.02.013

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

Barbier, E. B. (2017). Marine ecosystem services. In Current Biology (Vol. 27, Issue 11, pp. R507–R510). Cell Press. https://doi.org/10.1016/j.cub.2017.03.020

Barnes, D. K. A., Galgani, F., Thompson, R. C., & Barlaz, M. (2009). Accumulation and fragmentation of plastic debris in global environments. Philosophical Transactions of the Royal Society B: Biological Sciences, 364(1526), 1985–1998. https://doi.org/10.1098/rstb.2008.0205

Batel, A., Linti, F., Scherer, M., Erdinger, L., & Braunbeck, T. (2016). Transfer of benzo[a]pyrene from microplastics to Artemia nauplii and further to zebrafish via a trophic food web experiment: CYP1A induction and visual tracking of persistent organic pollutants. Environmental Toxicology and Chemistry, 35(7), 1656–1666. https://doi.org/10.1002/etc.3361

Bauer, J. E., Cai, W. J., Raymond, P. A., Bianchi, T. S., Hopkinson, C. S., & Regnier, P. A. G. (2013). The changing carbon cycle of the coastal ocean. In Nature (Vol. 504, Issue 7478, pp. 61–70). Nature Publishing Group. https://doi.org/10.1038/nature12857

Beach, Willis J. (1972) Skin Cleaner. U.S. Patent No. 3,645,904. Washington, DC: U.S. Patent and Trademark Office.

Bellas, J., & Gil, I. (2020). Polyethylene microplastics increase the toxicity of chlorpyrifos to the marine copepod Acartia tonsa. Environmental Pollution, 260, 114059. https://doi.org/10.1016/j.envpol.2020.114059

Berggreen, U., Hansen, B., & Kiørboe, T. (1988). Food size spectra, ingestion and growth of the copepod Acartia tonsa during development: Implications for determination of copepod production. Marine Biology, 99(3), 341–352. https://doi.org/10.1007/BF02112126

Bergmann, M., Mützel, S., Primpke, S., Tekman, M. B., Trachsel, J., & Gerdts, G. (2019). White and wonderful? Microplastics prevail in snow from the Alps to the Arctic. Science Advances, 5(8), eaax1157. https://doi.org/10.1126/sciadv.aax1157

Besseling, E., Foekema, E. M., Van Franeker, J. A., Leopold, M. F., Kühn, S., Bravo Rebolledo, E. L., Heße, E., Mielke, L., IJzer, J., Kamminga, P., & Koelmans, A. A. (2015). Microplastic in a macro filter feeder: Humpback whale Megaptera novaeangliae. Marine Pollution Bulletin, 95(1), 248–252. https://doi.org/10.1016/j.marpolbul.2015.04.007

Blastic. (2018). Toxicity of plastics. European Union. https://www.blastic.eu/knowledge-bank/impacts/toxicity-plastics/

Bonnet, D., Richardson, A., Harris, R., Hirst, A., Beaugrand, G., Edwards, M., Ceballos, S., Diekman, R., López-Urrutia, A., Valdes, L., Carlotti, F., Molinero, J. C., Weikert, H., Greve, W., Lucic, D., Albaina, A., Yahia, N. D., Umani, S. F., Miranda, A., … Fernandez De Puelles, M. L. (2005). An overview of Calanus helgolandicus ecology in European waters. In Progress in Oceanography (Vol. 65, Issue 1, pp. 1–53). Elsevier Ltd. https://doi.org/10.1016/j.pocean.2005.02.002

Bopp, L., Aumont, O., Cadule, P., Alvain, S., & Gehlen, M. (2005). Response of diatoms distribution to global warming and potential implications: A global model study. Geophysical Research Letters, 32(19), n/a-n/a. https://doi.org/10.1029/2005GL023653

Bopp, L., Bowler, C., Guidi, L., Karsenti, É., & de Vargas, C. (2020). The Ocean: A Carbon Pump. Ocean & Climate Platform. www.ocean-climate.org

Botterell, Z. L. R., Beaumont, N., Dorrington, T., Steinke, M., Thompson, R. C., & Lindeque, P. K. (2019). Bioavailability and effects of microplastics on marine zooplankton: A review. In Environmental Pollution (Vol. 245, pp. 98–110). Elsevier Ltd. https://doi.org/10.1016/j.envpol.2018.10.065

Boyd, P. W. (2015). Toward quantifying the response of the oceans' biological pump to climate change. Frontiers in Marine Science, 2(OCT), 77. https://doi.org/10.3389/fmars.2015.00077

Brandon, J. A., Freibott, A., & Sala, L. M. (2020). Patterns of suspended and salp-ingested microplastic debris in the North Pacific investigated with epifluorescence microscopy. Limnology and Oceanography Letters, 5(1), 46–53. https://doi.org/10.1002/lol2.10127

Browne, M. A., Galloway, T., & Thompson, R. (2007). Microplastic--an emerging contaminant of potential concern? Integrated Environmental Assessment and Management, 3(4), 559–561. https://doi.org/10.1897/1551-3793(2007)3[559:LD]2.0.CO;2

Browne, M. A., Dissanayake, A., Galloway, T. S., Lowe, D. M., & Thompson, R. C. (2008). Ingested microscopic plastic translocates to the circulatory system of the mussel, Mytilus edulis (L.). Environmental Science and Technology, 42(13), 5026–5031. https://doi.org/10.1021/es800249a

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

Bugoni, L., Krause, L., & Petry, M. V. (2001). Marine debris and human impacts on sea turtles in Southern Brazil. Marine Pollution Bulletin, 42(12), 1330–1334. https://doi.org/10.1016/S0025-326X(01)00147-3

CalRecycle. (2018). Methodology for Determining Remaining Landfill Capacity. California Department of Resources Recycling and Recovery. https://www.calrecycle.ca.gov/lea/advisories/45

Canniff, P. M., & Hoang, T. C. (2018). Microplastic ingestion by Daphnia magna and its enhancement on algal growth. Science of the Total Environment, 633, 500–507. https://doi.org/10.1016/j.scitotenv.2018.03.176

Castañeda, R. A., Avlijas, S., Simard, M. A., & Ricciardi, A. (2014). Microplastic pollution in St. Lawrence River sediments. *Canadian Journal of Fisheries and Aquatic Sciences*, *71*(12), 1767–1771. https://doi.org/10.1139/cjfas-2014-0281

Cau, A., Avio, C. G., Dessì, C., Follesa, M. C., Moccia, D., Regoli, F., & Pusceddu, A. (2019). Microplastics in the crustaceans Nephrops norvegicus and Aristeus antennatus: Flagship species for deep-sea environments? Environmental Pollution, 255. https://doi.org/10.1016/j.envpol.2019.113107

Cavan, E. L., Henson, S. A., Belcher, A., & Sanders, R. (2017). Role of zooplankton in determining the efficiency of the biological carbon pump. *Biogeosciences*, *14*, 177–186. https://doi.org/10.5194/bg-14-177-2017

Cavan, E. L., Laurenceau-Cornec, E. C., Bressac, M., & Boyd, P. W. (2019). Exploring the ecology of the mesopelagic biological pump. In Progress in Oceanography (Vol. 176, p. 102125). Elsevier Ltd. https://doi.org/10.1016/j.pocean.2019.102125

Charette, M. A.; Smith, W. H. The volume of Earth's ocean. Oceanography. 2010, 23, 112−114

Chiba, S., Saito, H., Fletcher, R., Yogi, T., Kayo, M., Miyagi, S., Ogido, M., & Fujikura, K. (2018). Human footprint in the abyss: 30 year records of deep-sea plastic debris. Marine Policy, 96, 204–212. https://doi.org/10.1016/j.marpol.2018.03.022

Choy, C. A., Robison, B. H., Gagne, T. O., Erwin, B., Firl, E., Halden, R. U., Hamilton, J. A., Katija, K., Lisin, S. E., Rolsky, C., & S. Van Houtan, K. (2019). The vertical distribution and biological transport of marine microplastics across the epipelagic and mesopelagic water column. Scientific Reports, 9(1). https://doi.org/10.1038/s41598-019-44117-2

Chubarenko, I., Bagaev, A., Zobkov, M., & Esiukova, E. (2016). On some physical and dynamical properties of microplastic particles in marine environment. Marine Pollution Bulletin, 108(1–2), 105–112. https://doi.org/10.1016/j.marpolbul.2016.04.048

Cole, M., Lindeque, P., Halsband, C., & Galloway, T. S. (2011). Microplastics as contaminants in the marine environment: A review. In Marine Pollution Bulletin (Vol. 62, Issue 12, pp. 2588–2597). https://doi.org/10.1016/j.marpolbul.2011.09.025

Cole, M., Lindeque, P., Fileman, E., Halsband, C., Goodhead, R., Moger, J., & Galloway, T. S. (2013). Microplastic ingestion by zooplankton. Environmental Science and Technology, 47(12), 6646–6655. https://doi.org/10.1021/es400663f

Cole, M., Lindeque, P. K., Fileman, E., Clark, J., Lewis, C., Halsband, C., & Galloway, T. S. (2016). Microplastics Alter the Properties and Sinking Rates of Zooplankton Faecal Pellets. Environmental Science and Technology, 50(6), 3239–3246. https://doi.org/10.1021/acs.est.5b05905

Cole, M., Coppock, R., Lindeque, P. K., Altin, D., Reed, S., Pond, D. W., Sørensen, L., Galloway, T. S., & Booth, A. M. (2019). Effects of Nylon Microplastic on Feeding, Lipid Accumulation, and Moulting in a Coldwater Copepod. Environmental Science and Technology, 53(12), 7075–7082. https://doi.org/10.1021/acs.est.9b01853

Coppock, R. L., Galloway, T. S., Cole, M., Fileman, E. S., Queirós, A. M., & Lindeque, P. K. (2019). Microplastics alter feeding selectivity and faecal density in the copepod, Calanus helgolandicus. Science of the Total Environment, 687, 780–789. https://doi.org/10.1016/j.scitotenv.2019.06.009

Corcoran, P. L., Biesinger, M. C., & Grifi, M. (2009). Plastics and beaches: A degrading relationship. Marine Pollution Bulletin, 58(1), 80–84. https://doi.org/10.1016/j.marpolbul.2008.08.022

Cózar, A., Echevarría, F., González-Gordillo, J. I., Irigoien, X., Úbeda, B., Hernández-León, S., Palma, Á. T., Navarro, S., García-de-Lomas, J., Ruiz, A., Fernández-de-Puelles, M. L., & Duarte, C. M. (2014). Plastic debris in the open ocean. Proceedings of the National Academy of Sciences of the United States of America, 111(28), 10239–10244. https://doi.org/10.1073/pnas.1314705111

Cressey, D. (2016). Bottles, bags, ropes and toothbrushes: The struggle to track ocean plastics. Nature, 536(7616), 263–265. https://doi.org/10.1038/536263a

Derraik, J. G. B. (2002). The pollution of the marine environment by plastic debris: A review. In Marine Pollution Bulletin (Vol. 44, Issue 9, pp. 842–852). https://doi.org/10.1016/S0025-326X(02)00220-5

De Stephanis, R., Giménez, J., Carpinelli, E., Gutierrez-Exposito, C., & Cañadas, A. (2013). As main meal for sperm whales: Plastics debris. Marine Pollution Bulletin, 69(1–2), 206–214. https://doi.org/10.1016/j.marpolbul.2013.01.033

Devriese, L. I., van der Meulen, M. D., Maes, T., Bekaert, K., Paul-Pont, I., Frère, L., Robbens, J., & Vethaak, A. D. (2015). Microplastic contamination in brown shrimp (Crangon crangon, Linnaeus 1758) from coastal waters of the Southern North Sea and Channel area. Marine Pollution Bulletin, 98(1–2), 179–187. https://doi.org/10.1016/j.marpolbul.2015.06.051

DG Environment Report. (2017). Intentionally added microplastics in products - Final report. European Commission (DG Environment). Amec Foster Wheeler Environment & Infrastructure UK Limited, October, 1–220. http://ec.europa.eu/environment/chemicals/reach/pdf/39168 Intentionally added microplastics - Final report 20171020.pdf

Doyle, M. J., Watson, W., Bowlin, N. M., & Sheavly, S. B. (2011). Plastic particles in coastal pelagic ecosystems of the Northeast Pacific ocean. Marine Environmental Research, 71(1), 41–52. https://doi.org/10.1016/j.marenvres.2010.10.001

Ducklow, H. W., Steinberg, D. K., & Buesseler, K. O. (2001). Upper Ocean Carbon Export and the Biological Pump. Oceanography, 14(4).

Duis, K., & Coors, A. (2016). Microplastics in the aquatic and terrestrial environment: sources (with a specific focus on personal care products), fate and effects. In Environmental Sciences Europe (Vol. 28, Issue 1, pp. 1–25). Springer Verlag. https://doi.org/10.1186/s12302-015-0069-y

Emerson, C. E., Reinardy, H. C., Bates, N. R., & Bodnar, A. G. (2017). Ocean acidification impacts spine integrity but not regenerative capacity of spines and tube feet in adult sea urchins. Royal Society Open Science, 4(5). https://doi.org/10.1098/rsos.170140

Eriksen, M., Maximenko, N., Thiel, M., Cummins, A., Lattin, G., Wilson, S., Hafner, J., Zellers, A., & Rifman, S. (2013). Plastic pollution in the South Pacific subtropical gyre. Marine Pollution Bulletin, 68(1–2), 71–76. https://doi.org/10.1016/j.marpolbul.2012.12.021

Eriksen, M., Lebreton, L. C. M., Carson, H. S., Thiel, M., Moore, C. J., Borerro, J. C., Galgani, F., Ryan, P. G., & Reisser, J. (2014). Plastic Pollution in the World's Oceans: More than 5 Trillion Plastic Pieces Weighing over 250,000 Tons Afloat at Sea. PLoS ONE, 9(12), e111913. https://doi.org/10.1371/journal.pone.0111913

Erni-Cassola, G., Zadjelovic, V., Gibson, M. I., & Christie-Oleza, J. A. (2019). Distribution of plastic polymer types in the marine environment; A meta-analysis. Journal of Hazardous Materials, 369, 691–698. https://doi.org/10.1016/j.jhazmat.2019.02.067

Farrell, P., & Nelson, K. (2013). Trophic level transfer of microplastic: Mytilus edulis (L.) to Carcinus maenas (L.). Environmental Pollution, 177, 1–3. https://doi.org/10.1016/j.envpol.2013.01.046

FDA U.S. FOOD & DRUG ADMINISTRATION. (2015). The Microbead-Free Waters Act: FAQs | FDA. https://www.fda.gov/cosmetics/cosmetics-laws-regulations/microbead-free-waters-act-faqs

Fendall, L. S., & Sewell, M. A. (2009). Contributing to marine pollution by washing your face: Microplastics in facial cleansers. Marine Pollution Bulletin, 58(8), 1225–1228. https://doi.org/10.1016/j.marpolbul.2009.04.025

Foley, C. J., Feiner, Z. S., Malinich, T. D., & Höök, T. O. (2018). A meta-analysis of the effects of exposure to microplastics on fish and aquatic invertebrates. Science of the Total Environment, 631–632, 550–559. https://doi.org/10.1016/j.scitotenv.2018.03.046

Folger, P. (2009). The carbon cycle: Implications for climate change and congress. In Carbon Capture and Storage including Coal-Fired Power Plants (pp. 99–111).

Frankenberg, D., & Smith, K. L. (1967). COPROPHAGY IN MARINE ANIMALS. Limnology and Oceanography, 12(3), 443–450. https://doi.org/10.4319/lo.1967.12.3.0443

Fraser, C. I., Morrison, A. K., Hogg, A. M. C., Macaya, E. C., van Sebille, E., Ryan, P. G., Padovan, A., Jack, C., Valdivia, N., & Waters, J. M. (2018). Antarctica's ecological isolation will be broken by storm-driven dispersal and warming. In Nature Climate Change (Vol. 8, Issue 8, pp. 704–708). Nature Publishing Group. https://doi.org/10.1038/s41558-018-0209-7

Fry, D. M., Fefer, S. I., & Sileo, L. (1987). Ingestion of plastic debris by Laysan Albatrosses and Wedge-tailed Shearwaters in the Hawaiian Islands. Marine Pollution Bulletin, 18(6 SUPPL. B), 339–343. https://doi.org/10.1016/S0025-326X(87)80022-X

Galafassi, S., Nizzetto, L., & Volta, P. (2019). Plastic sources: A survey across scientific and grey literature for their inventory and relative contribution to microplastics pollution in natural environments, with an emphasis on surface water. In Science of the Total Environment (Vol. 693, p. 133499). Elsevier B.V. https://doi.org/10.1016/j.scitotenv.2019.07.305

Geilfus, N. X., Munson, K. M., Sousa, J., Germanov, Y., Bhugaloo, S., Babb, D., & Wang, F. (2019). Distribution and impacts of microplastic incorporation within sea ice. Marine Pollution Bulletin, 145, 463–473. https://doi.org/10.1016/j.marpolbul.2019.06.029

Geyer, R., Jambeck, J. R., & Law, K. L. (2017). Production, use, and fate of all plastics ever made. Science Advances, 3(7). https://doi.org/10.1126/sciadv.17007822.

Goldstein, M. C., & Goodwin, D. S. (2013). Gooseneck barnacles (Lepas spp.) ingest microplastic debris in the north Pacific subtropical gyre. PeerJ, 2013(1). https://doi.org/10.7717/peerj.184

Gonzalez, G. (2014). ADW: Acartia tonsa: INFORMATION. Animal Diversity Web. https://animaldiversity.org/accounts/Acartia_tonsa/

Grand View Research. (2020). Traffic Road Marking Coatings Market Size | Industry Report, 2019-2025. Grand Review Research. https://www.grandviewresearch.com/industry-analysis/traffic-road-marking-coatings-market

Green, D. S. (2016). Effects of microplastics on European flat oysters, Ostrea edulis and their associated benthic communities. Environmental Pollution, 216, 95–103. https://doi.org/10.1016/j.envpol.2016.05.043

Gregory, M. R. (1991). The hazards of persistent marine pollution: Drift plastics and conservation islands. Journal of the Royal Society of New Zealand, 21(2), 83–100. https://doi.org/10.1080/03036758.1991.10431398

Greve, W. (1977). Interspecific interaction: The analysis of complex structures in carnivorous zooplankton populations. Helgoländer Wissenschaftliche Meeresuntersuchungen, 30(1–4), 83–91. https://doi.org/10.1007/BF02207827

Hain, M. P., Sigman, D. M., & Haug, G. H. (2013). The Biological Pump in the Past. In Treatise on Geochemistry: Second Edition (Vol. 8, pp. 485–517). Elsevier Inc. https://doi.org/10.1016/B978-0-08-095975-7.00618-5

Hall, N. M., Berry, K. L. E., Rintoul, L., & Hoogenboom, M. O. (2015). Microplastic ingestion by scleractinian corals. Marine Biology, 162(3), 725–732. https://doi.org/10.1007/s00227-015-2619-7

Hammer, A., Grüttner, C., & Schumann, R. (1999). The effect of electrostatic charge of food particles on capture efficiency by Oxyrrhis marina Dujardin (Dinoflagellate). Protist, 150(4), 375–382. https://doi.org/10.1016/S1434-4610(99)70039-8

Hammer, J., Kraak, M. H. S., & Parsons, J. R. (2012). Plastics in the marine environment: The dark side of a modern gift. In Reviews of Environmental Contamination and Toxicology (Vol. 220, pp. 1–44). Springer, New York, NY. https://doi.org/10.1007/978-1-4614-3414-6_1

Hammill, E., Johnson, E., Atwood, T. B., Harianto, J., Hinchliffe, C., Calosi, P., & Byrne, M. (2018). Ocean acidification alters zooplankton communities and increases top-down pressure of a cubozoan predator. Global Change Biology, 24(1), e128–e138. https://doi.org/10.1111/gcb.13849

Hansell, D. A. (2002). DOC in the Global Ocean Carbon Cycle. In Biogeochemistry of Marine Dissolved Organic Matter (pp. 685–715). Academic Press. https://doi.org/10.1016/b978-012323841-2/50017-8

Hansen, A. N., & Visser, A. W. (2016). Carbon export by vertically migrating zooplankton: An optimal behavior model. Limnology and Oceanography, 61(2), 701–710. https://doi.org/10.1002/lno.10249

Hidalgo-Ruz, V., Gutow, L., Thompson, R. C., & Thiel, M. (2012). Microplastics in the marine environment: A review of the methods used for identification and quantification. Environmental Science and Technology, 46(6), 3060–3075. https://doi.org/10.1021/es2031505

Hinojosa, I. A., & Thiel, M. (2009). Floating marine debris in fjords, gulfs and channels of southern Chile. Marine Pollution Bulletin, 58(3), 341–350. https://doi.org/10.1016/j.marpolbul.2008.10.020

Honjo, S., Eglinton, T., Taylor, C., Ulmer, K., Sievert, S., Bracher, A., German, C., Edgcomb, V., Francois, R., Iglesias-Rodriguez, M. D., Van Mooy, B., & Rapeta, D. (2014). Understanding the Role of the Biological

Hopewell, J., Dvorak, R., & Kosior, E. (2009). Plastics recycling: Challenges and opportunities. In Philosophical Transactions of the Royal Society B: Biological Sciences (Vol. 364, Issue 1526, pp. 2115–2126). Royal Society. https://doi.org/10.1098/rstb.2008.0311

Hui, M., Shengyan, P., Shibin, L., Yingchen, B., Mandal, S., & Baoshan, X. (2020). Microplastics in aquatic environments: Toxicity to trigger ecological consequences. Environmental Pollution, 114089. https://doi.org/10.1016/j.envpol.2020.114089

Iannilli, V., Di Gennaro, A., Lecce, F., Sighicelli, M., Falconieri, M., Pietrelli, L., Poeta, G., & Battisti, C. (2018). Microplastics in Talitrus saltator (Crustacea, Amphipoda): new evidence of ingestion from natural contexts. Environmental Science and Pollution Research, 25(28), 28725–28729. https://doi.org/10.1007/s11356-018-2932-z

IUCN. (2015). Plastic debris in the ocean: the characterization of marine plastics and their environmental impacts, situation analysis report. In Plastic debris in the ocean: the characterization of marine plastics and their environmental impacts, situation analysis report. International Union for Conservation of Nature. https://doi.org/10.2305/iucn.ch.2014.03.en

IUCN. (2018). Marine Plastics. In IUCN Issues Brief. https://www.iucn.org/resources/issues-briefs/marine-plastics

Jacobsen, J. K., Massey, L., & Gulland, F. (2010). Fatal ingestion of floating net debris by two sperm whales (Physeter macrocephalus). Marine Pollution Bulletin, 60(5), 765–767. https://doi.org/10.1016/j.marpolbul.2010.03.008

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

Jamieson, A. J., Brooks, L. S. R., Reid, W. D. K., Piertney, S. B., Narayanaswamy, B. E., & Linley, T. D. (2019). Microplastics and synthetic particles ingested by deep-sea amphipods in six of the deepest marine ecosystems on Earth. Royal Society Open Science, 6(2). https://doi.org/10.1098/rsos.180667

Jeong, C. B., Kang, H. M., Lee, M. C., Kim, D. H., Han, J., Hwang, D. S., Souissi, S., Lee, S. J., Shin, K. H., Park, H. G., & Lee, J. S. (2017). Adverse effects of microplastics and oxidative stress-induced MAPK/Nrf2 pathway-mediated defense mechanisms in the marine copepod Paracyclopina nana. Scientific Reports, 7(1), 1–11. https://doi.org/10.1038/srep41323

Jonsson, P., & Tiselius, P. (1990). Feeding behaviour, prey detection and capture efficiency of the copepod Acartia tonsa feeding on planktonic ciliates. Marine Ecology Progress Series, 60, 35–44. https://doi.org/10.3354/meps060035

Kiørboe, T., Møhlenberg, F., & Hamburger, K. (1985). Bioenergetics of the planktonic copepod Acartia tonsa: relation between feeding, egg production and respiration, and composition of specific dynamic action. Marine Ecology Progress Series, 26, 85–97. https://doi.org/10.3354/meps026085

Kiorboe, T. (2008). A mechanistic approach to plankton ecology. Princeton University.

Kiørboe, T. (2011). How zooplankton feed: mechanisms, traits and trade-offs. Biological Reviews, 86(2), 311–339. https://doi.org/10.1111/j.1469-185X.2010.00148.x

Knutson, T., Camargo, S. J., Chan, J. C. L., Emanuel, K., Ho, C. H., Kossin, J., Mohapatra, M., Satoh, M., Sugi, M., Walsh, K., & Wu, L. (2019). Tropical cyclones and climate change assessment. Bulletin of the American Meteorological Society, 100(10), 1987–2007. https://doi.org/10.1175/BAMS-D-18-0189.1

Komar, P. D., Morse, A. P., Small, L. F., & Fowler, S. W. (1981). An analysis of sinking rates of natural copepod and euphausiid fecal pellets. *Limnology and Oceanography*, *26*(1), 172–180. https://doi.org/10.4319/lo.1981.26.1.0172

Kuijper, Lothar D. J., Anderson, Thomas R., & Kooijman, Sebastiaan. A. L. M. (2004). C and N gross growth efficiencies of copepod egg production studied using a Dynamic Energy Budget model. Journal of Plankton Research, 26(2), 213–226. https://doi.org/10.1093/plankt/fbh020

Kubota, M. (1994). A mechanism for the accumulation of floating marine debris north of Hawaii. Journal of Physical Oceanography, 24(5), 1059–1064. https://doi.org/10.1175/1520-0485(1994)024<1059:AMFTAO>2.0.CO;2

Kubota, M., Takayama, K., & Namimoto, D. (2005). Pleading for the use of biodegradable polymers in favor of marine environments and to avoid an asbestos-like problem for the future. In Applied Microbiology and Biotechnology (Vol. 67, Issue 4, pp. 469–476). Springer. https://doi.org/10.1007/s00253-004-1857-2

Kurihara, H., & Shirayama, Y. (2004). Effects of increased atmospheric CO2 on sea urchin early development. Marine Ecology Progress Series, 274, 161–169. https://doi.org/10.3354/meps274161

Laffoley, D. & Baxter, J. M. (editors). 2016. Explaining ocean warming: Causes, scale, effects and consequences. Full report. Gland, Switzerland: IUCN. 456 pp. https://doi.org/10.2305/IUCN.CH.2016.08.en

Lattin, G. L., Moore, C. J., Zellers, A. F., Moore, S. L., & Weisberg, S. B. (2004). A comparison of neustonic plastic and zooplankton at different depths near the southern California shore. Marine Pollution Bulletin, 49(4), 291–294. https://doi.org/10.1016/j.marpolbul.2004.01.020

Lassen, C., Hansen, S. F., Magnusson, K., Norén, F., Hartmann, N. I. B., Jensen, P. R., Nielsen, T. G., & Brinch, A. (2015). Microplastics - Occurrence, effects and sources of releases to the environment in Denmark. In Danish Environmental Protection Agency. (Issue 1793, pp. 33-35,45-46,82-84,120-141). Danish Environmental Protection Agency. www.eng.mst.dk

Law, K. L., & Thompson, R. C. (2014). Microplastics in the seas. In Science (Vol. 345, Issue 6193, pp. 144–145). American Association for the Advancement of Science. https://doi.org/10.1126/science.1254065

Leandro, S. M., Tiselius, P., & Queiroga, H. (2006). Growth and development of nauplii and copepodites of the estuarine copepod Acartia tonsa from southern Europe (Ria de Aveiro, Portugal) under saturating food conditions. Marine Biology, 150(1), 121–129. https://doi.org/10.1007/s00227-006-0336-y

Lee, K. W., Shim, W. J., Kwon, O. Y., & Kang, J. H. (2013). Size-dependent effects of micro polystyrene particles in the marine copepod tigriopus japonicus. Environmental Science and Technology, 47(19), 11278–11283. https://doi.org/10.1021/es401932b

Le Guern, C. (2009). Plastic Pollution. Santa Aguila Foundation. https://plastic-pollution.org/

Le Quéré, C., Takahashi, T., Buitenhuis, E. T., Rödenbeck, C., & Sutherland, S. C. (2010). Impact of climate change and variability on the global oceanic sink of CO2. Global Biogeochemical Cycles, 24(4), n/a-n/a. https://doi.org/10.1029/2009GB003599

León, V. M., García, I., González, E., Samper, R., Fernández-González, V., & Muniategui-Lorenzo, S. (2018). Potential transfer of organic pollutants from littoral plastics debris to the marine environment. Environmental Pollution, 236, 442–453. https://doi.org/10.1016/j.envpol.2018.01.114

Leslie, H. A. (2017). Review of Microplastics in Cosmetics Scientific background on a potential source of plastic particulate marine litter to support decision-making. IVM Institute for Environmental Studies, 1–33. http://www.ivm.vu.nl/en/Images/Plastic_ingredients_in_Cosmetics_07-2014_FINAL_tcm234-409859.pdf%0Ahttp://aviationweek.com/awincommercial/safety-recommendations-go-unanswered-indonesia

Lo, H. K. A., & Chan, K. Y. K. (2018). Negative effects of microplastic exposure on growth and development of Crepidula onyx. Environmental Pollution, 233, 588–595. https://doi.org/10.1016/j.envpol.2017.10.095

Lobelle, D., & Cunliffe, M. (2011). Early microbial biofilm formation on marine plastic debris. Marine Pollution Bulletin, 62(1), 197–200. https://doi.org/10.1016/j.marpolbul.2010.10.013

Long, M., Moriceau, B., Gallinari, M., Lambert, C., Huvet, A., Raffray, J., & Soudant, P. (2015). Interactions between microplastics and phytoplankton aggregates: Impact on their respective fates. Marine Chemistry, 175, 39–46. https://doi.org/10.1016/j.marchem.2015.04.003

Longhurst, A. R., & Glen Harrison, W. (1988). Vertical nitrogen flux from the oceanic photic zone by diel migrant zooplankton and nekton. Deep Sea Research Part A, Oceanographic Research Papers, 35(6), 881–889. https://doi.org/10.1016/0198-0149(88)90065-9

Longhurst, A. R., & Glen Harrison, W. (1989). The biological pump: Profiles of plankton production and consumption in the upper ocean. Progress in Oceanography, 22(1), 47–123. https://doi.org/10.1016/0079-6611(89)90010-4

Longhurst, A. R., Bedo, A. W., Harrison, W. G., Head, E. J. H., & Sameoto, D. D. (1990). Vertical flux of respiratory carbon by oceanic diel migrant biota. Deep Sea Research Part A, Oceanographic Research Papers, 37(4), 685–694. https://doi.org/10.1016/0198-0149(90)90098-G

Lusher, A. L., Tirelli, V., O'Connor, I., & Officer, R. (2015). Microplastics in Arctic polar waters: The first reported values of particles in surface and sub-surface samples. Scientific Reports, 5. https://doi.org/10.1038/srep14947

Macfadyen, G., Huntington, T., & Cappell, R. (2009). Abandoned, lost or otherwise discarded fishing gear. In FAO Fisheries and Aquaculture Technical Paper 523 (Vol. 523). http://www.unep.org/regionalseas/marinelitter/publications/default.asp

Mackey, A. P., Atkinson, A., Hill, S. L., Ward, P., Cunningham, N. J., Johnston, N. M., & Murphy, E. J. (2012). Antarctic macrozooplankton of the southwest Atlantic sector and Bellingshausen Sea: Baseline historical distributions (Discovery Investigations, 1928-1935) related to temperature and food, with projections for subsequent ocean warming. Deep-Sea Research Part II: Topical Studies in Oceanography, 59–60, 130–146. https://doi.org/10.1016/j.dsr2.2011.08.011

Magni, S., Binelli, A., Pittura, L., Avio, C. G., Della Torre, C., Parenti, C. C., Gorbi, S., & Regoli, F. (2019). The fate of microplastics in an Italian Wastewater Treatment Plant. Science of the Total Environment, 652, 602–610. https://doi.org/10.1016/j.scitotenv.2018.10.269

Magnusson, K., Eliasson, K., Fråne, A., Haikonen, K., Hultén, J., Olshammar, M., Stadmark, J., & Voisin, A. (2016). Swedish sources and pathways for microplastics to the marine environment. A review of existing data. IVL Svenska Miljöinstitutet, C 183, 1–89. www.ivl.se

Mahon, A. M., O'Connell, B., Healy, M. G., O'Connor, I., Officer, R., Nash, R., & Morrison, L. (2017). Microplastics in sewage sludge: Effects of treatment. Environmental Science and Technology, 51(2), 810–818. https://doi.org/10.1021/acs.est.6b04048

Marcus, N. H., & Wilcox, J. A. (2007). A GUIDE TO THE MESO-SCALE PRODUCTION OF THE COPEPOD ACARTIA TONSA.

Marcus, N. H. (2011). Zooplankton: Responses to and consequences of hypoxia. In Coastal Hypoxia: Consequences for Living Resources and Ecosystems (pp. 49–60). American Geophysical Union (AGU). https://doi.org/10.1029/ce058p0049

Martinez, E., Maamaatuaiahutapu, K., & Taillandier, V. (2009). Floating marine debris surface drift: Convergence and accumulation toward the South Pacific subtropical gyre. Marine Pollution Bulletin, 58(9), 1347–1355. https://doi.org/10.1016/j.marpolbul.2009.04.022

Mathalon, A., & Hill, P. (2014). 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

Mauchline, J. (1998). Advances in Marine Biology, The Biology of Calanoid Copepods (J. H. S. Blaxter, A. J. Southward, & P. A. Tyler (eds.); Vol. 33). Academic Press. https://doi.org/10.1016/s0065-2881(08)60230-8

McManus, M. A., & Woodson, C. B. (2012). Plankton distribution and ocean dispersal. Journal of Experimental Biology, 215(6), 1008–1016. https://doi.org/10.1242/jeb.059014

Md Amin, R., Sohaimi, E. S., Anuar, S. T., & Bachok, Z. (2020). Microplastic ingestion by zooplankton in Terengganu coastal waters, southern South China Sea. Marine Pollution Bulletin, 150, 110616. https://doi.org/10.1016/j.marpolbul.2019.110616

Miljødirektoratet (Agência Norueguesa do Ambiente). (2014). Sources of microplastic-pollution to the marine environment - Relatório de Missão M-321. https://www.miljodirektoratet.no/globalassets/publikasjoner/M321/M321.pdf

Miranda, D. de A., & de Carvalho-Souza, G. F. (2016). Are we eating plastic-ingesting fish? Marine Pollution Bulletin, 103(1–2), 109–114. https://doi.org/10.1016/j.marpolbul.2015.12.035

Moore, C. J., Moore, S. L., Leecaster, M. K., & Weisberg, S. B. (2001). A comparison of plastic and plankton in the North Pacific Central Gyre. Marine Pollution Bulletin, 42(12), 1297–1300. https://doi.org/10.1016/S0025-326X(01)00114-X

Moore, C. J. (2008). Synthetic polymers in the marine environment: A rapidly increasing, long-term threat. Environmental Research, 108(2), 131–139. https://doi.org/10.1016/j.envres.2008.07.025

Morán, X. A. G., López-Urrutia, Á., Calvo-Díaz, A., & Li, W. K. W. (2010). Increasing importance of small phytoplankton in a warmer ocean. Global Change Biology, 16(3), 1137–1144. https://doi.org/10.1111/j.1365-2486.2009.01960.x

Morét-Ferguson, S., Law, K. L., Proskurowski, G., Murphy, E. K., Peacock, E. E., & Reddy, C. M. (2010). The size, mass, and composition of plastic debris in the western North Atlantic Ocean. Marine Pollution Bulletin, 60(10), 1873–1878. https://doi.org/10.1016/j.marpolbul.2010.07.020

Murray, F., & Cowie, P. R. (2011). Plastic contamination in the decapod crustacean Nephrops norvegicus (Linnaeus, 1758). Marine Pollution Bulletin, 62(6), 1207–1217. https://doi.org/10.1016/j.marpolbul.2011.03.032

Napper, I. E., & Thompson, R. C. (2016). Release of synthetic microplastic plastic fibres from domestic washing machines: Effects of fabric type and washing conditions. Marine Pollution Bulletin, 112(1–2), 39–45. https://doi.org/10.1016/j.marpolbul.2016.09.025

NASA. (2010). NASA - A Big Net Gain. In Spinoff. Brian Dunbar. http://www.nasa.gov/offices/oct/home/tech_life_buoy.html

Nelms, S. E., Galloway, T. S., Godley, B. J., Jarvis, D. S., & Lindeque, P. K. (2018). Investigating microplastic trophic transfer in marine top predators. Environmental Pollution, 238, 999–1007. https://doi.org/10.1016/j.envpol.2018.02.016

Nelms, S. E., Barnett, J., Brownlow, A., Davison, N. J., Deaville, R., Galloway, T. S., Lindeque, P. K., Santillo, D., & Godley, B. J. (2019). Microplastics in marine mammals stranded around the British coast: ubiquitous but transitory? Scientific Reports, 9(1), 1–8. https://doi.org/10.1038/s41598-018-37428-3

NOAA. (2020a). What We Know About "Ghost Fishing." Marine Debris Program. https://marinedebris.noaa.gov/what-we-know-about-ghost-fishing-0

NOAA. (2020b). Why should we care about the ocean? National Ocean Service website, https://oceanservice.noaa.gov/facts/why-care-about-ocean.html

Nobili, R., Robinson, C., Buitenhuis, E., & Castellani, C. (2013). Food quality regulates the metabolism and reproduction of Temora longicornis. Biogeosciences Discuss, 10, 3203–3239. https://doi.org/10.5194/bgd-10-3203-2013

Norrbin, M. F. (1994). Seasonal patterns in gonad maturation, sex ratio and size in some small, high-latitude copepods: implications for overwintering tactics. Journal of Plankton Research, 16(2), 115–131. https://doi.org/10.1093/plankt/16.5.581-a

Onink, V., Wichmann, D., Delandmeter, P., & van Sebille, E. (2019). The Role of Ekman Currents, Geostrophy, and Stokes Drift in the Accumulation of Floating Microplastic. Journal of Geophysical Research: Oceans, 124(3), 1474–1490. https://doi.org/10.1029/2018JC014547

OSPAR. (2009). Marine litter in the North-East Atlantic Region: Assessment and priorities for response. London, United Kingdom, 127 pp.

Ottvall, L.-O., & Carlsson, P. (2016). The Impact of Polystyrene Microplastics on Filtration Rate in the Marine Copepod Acartia tonsa.

Pachkowski, B. (2016). Microplastics as Contaminants of Emerging Concern. http://www.nist.gov/pml/wmd/metric/upload/size-and-scale-sesson.pdf

Paffenhöfer, G.-A., Bathmann, U., Bundy, M. H., Clarke, M. E., Cowles, T. J., Daly, K., Dam, H. G., Dekshenieks, M. M., Donaghay, P. L., Gibson, D. M., Gifford, D. J., Hansen, B. W., Hartline, D. K., Head, E. J. H., Hofmann, E. E., Hopcroft, R. R., Jahnke, R. A., Jonasdottir, S. H., Kiorboe, T., Kleppel, G. S., … Wishner, K. F. (2001). Future marine zooplankton research - a perspective. Marine Ecology Progress Series, 222, 297–308.

Passow, U., & Carlson, C. A. (2012). The biological pump in a high CO2 world. Marine Ecology Progress Series, 470, 249–271. https://doi.org/10.3354/meps09985

Peeken, I., Primpke, S., Beyer, B., Gütermann, J., Katlein, C., Krumpen, T., Bergmann, M., Hehemann, L., & Gerdts, G. (2018). Arctic sea ice is an important temporal sink and means of transport for microplastic. Nature Communications, 9(1), 1–12. https://doi.org/10.1038/s41467-018-03825-5

Pomerleau, C., Sastri, A. R., & Beisner, B. E. (2015). Evaluation of functional trait diversity for marine zooplankton communities in the Northeast subarctic Pacific Ocean. Journal of Plankton Research, 37(4), 712–726. https://doi.org/10.1093/plankt/fbv045

Prata, J. C., da Costa, J. P., Lopes, I., Duarte, A. C., & Rocha-Santos, T. (2019). Effects of microplastics on microalgae populations: A critical review. In Science of the Total Environment (Vol. 665, pp. 400–405). Elsevier B.V. https://doi.org/10.1016/j.scitotenv.2019.02.132

Prog, S., & Hirche, H.-J. (1983). MARINE ECOLOGY-PROGRESS SERIES Overwintering of Calanus finmarchicus and Calanus helgolandicus (Vol. 11). https://about.jstor.org/terms

Provencher, J. F., Bond, A. L., Hedd, A., Montevecchi, W. A., Muzaffar, S. Bin, Courchesne, S. J., Gilchrist, H. G., Jamieson, S. E., Merkel, F. R., Falk, K., Durinck, J., & Mallory, M. L. (2014). Prevalence of marine debris in marine birds from the North Atlantic. Marine Pollution Bulletin, 84(1–2), 411–417. https://doi.org/10.1016/j.marpolbul.2014.04.044

Reid, J. W., & Williamson, C. E. (2010). Copepoda. In Ecology and Classification of North American Freshwater Invertebrates (pp. 829–899). Elsevier Inc. https://doi.org/10.1016/B978-0-12-374855-3.00021-2

Richardson, A. J. (2008). In hot water: Zooplankton and climate change. ICES Journal of Marine Science, 65(3), 279–295. https://doi.org/10.1093/icesjms/fsn028

Rios, L. M., Moore, C., & Jones, P. R. (2007). Persistent organic pollutants carried by synthetic polymers in the ocean environment. Marine Pollution Bulletin, 54(8), 1230–1237. https://doi.org/10.1016/j.marpolbul.2007.03.022

Rochman, C. M., Kurobe, T., Flores, I., & Teh, S. J. (2014). Early warning signs of endocrine disruption in adult fish from the ingestion of polyethylene with and without sorbed chemical pollutants from the marine environment. Science of the Total Environment, 493, 656–661. https://doi.org/10.1016/j.scitotenv.2014.06.051

Rochman, C. M., Tahir, A., Williams, S. L., Baxa, D. V., Lam, R., Miller, J. T., Teh, F. C., Werorilangi, S., & Teh, S. J. (2015). Anthropogenic debris in seafood: Plastic debris and fibers from textiles in fish and bivalves sold for human consumption. Scientific Reports, 5(1), 1–10. https://doi.org/10.1038/srep14340

Rogers, D. C., & Thorp, J. H. (2015). Collecting, Preserving, and Culturing Invertebrates. In Thorp and Covich's Freshwater Invertebrates: Ecology and General Biology: Fourth Edition (Vol. 1, pp. 57–62). Elsevier Inc. https://doi.org/10.1016/B978-0-12-385026-3.00003-6

Roman, J., & McCarthy, J. J. (2010). The Whale Pump: Marine Mammals Enhance Primary Productivity in a Coastal Basin. PLoS ONE, 5(10), e13255. https://doi.org/10.1371/journal.pone.0013255

Roman, J., Estes, J. A., Morissette, L., Smith, C., Costa, D., McCarthy, J., Nation, J. B., Nicol, S., Pershing, A., & Smetacek, V. (2014). Whales as marine ecosystem engineers. In Frontiers in Ecology and the Environment (Vol. 12, Issue 7, pp. 377–385). Ecological Society of America. https://doi.org/10.1890/130220

Ryan, P. G., Moore, C. J., Van Franeker, J. A., & Moloney, C. L. (2009). Monitoring the abundance of plastic debris in the marine environment. In Philosophical Transactions of the Royal Society B: Biological Sciences (Vol. 364, Issue 1526, pp. 1999–2012). Royal Society. https://doi.org/10.1098/rstb.2008.0207

SEA. (2012). Plastic debris in the Ocean. Plastics at SEA North Pacific Expedition - Investigating the Effects of Plastic in the Ocean Ecosystem | Home. https://www.sea.edu/plastics/frequently_asked_questions

Sell, A. F., van Keuren, D., & Madin, L. P. (2001). Predation by omnivorous copepods on early developmental stages of Calanus finmarchicus and Pseudocalanus spp. Limnology and Oceanography, 46(4), 953–959. https://doi.org/10.4319/lo.2001.46.4.0953

Seltenrich, N. (2015). New link in the food chain? Marine plastic pollution and seafood safety. In Environmental Health Perspectives (Vol. 123, Issue 2, pp. A34–A41). Public Health Services, US Dept of Health and Human Services. https://doi.org/10.1289/ehp.123-A34

Seng, N., Lai, S., Fong, J., Saleh, M. F., Cheng, C., Cheok, Z. Y., & Todd, P. A. (2020). Early evidence of microplastics on seagrass and macroalgae. Marine and Freshwater Research. https://doi.org/10.1071/MF19177

Setälä, O., Fleming-Lehtinen, V., & Lehtiniemi, M. (2014). Ingestion and transfer of microplastics in the planktonic food web. Environmental Pollution, 185, 77–83. https://doi.org/10.1016/j.envpol.2013.10.013

Setälä, O., Lehtiniemi, M., Coppock, R., & Cole, M. (2018). Microplastics in Marine Food Webs. In Microplastic Contamination in Aquatic Environments (pp. 339–363). Elsevier. https://doi.org/10.1016/b978-0-12-813747-5.00011-4

Sharma, S., & Chatterjee, S. (2017). Microplastic pollution, a threat to marine ecosystem and human health: a short review. Environmental Science and Pollution Research, 24(27), 21530–21547. https://doi.org/10.1007/s11356-017-9910-8

Shim, W. J., & Thomposon, R. C. (2015). Microplastics in the Ocean. Archives of Environmental Contamination and Toxicology, 69(3), 265–268. https://doi.org/10.1007/s00244-015-0216-x

Smith, J. N., De'Ath, G., Richter, C., Cornils, A., Hall-Spencer, J. M., & Fabricius, K. E. (2016). Ocean acidification reduces demersal zooplankton that reside in tropical coral reefs. Nature Climate Change, 6(12), 1124–1129. https://doi.org/10.1038/nclimate3122

Steinberg, D. K., & Landry, M. R. (2017). Zooplankton and the Ocean Carbon Cycle. Annual Review of Marine Science, 9(1), 413–444. https://doi.org/10.1146/annurev-marine-010814-015924

Sun, X., Li, Q., Zhu, M., Liang, J., Zheng, S., & Zhao, Y. (2017). Ingestion of microplastics by natural zooplankton groups in the northern South China Sea. Marine Pollution Bulletin, 115(1–2), 217–224. https://doi.org/10.1016/j.marpolbul.2016.12.004

Sundt, P., Schultze, P., Syversen, F. (2014). Sources of microplastic- pollution to the marine environment. In Mepex, Norwegian Environment Agency. https://doi.org/M-321|2015

Sutherland, W. J., Clout, M., Côté, I. M., Daszak, P., Depledge, M. H., Fellman, L., Fleishman, E., Garthwaite, R., Gibbons, D. W., De Lurio, J., Impey, A. J., Lickorish, F., Lindenmayer, D., Madgwick, J., Margerison, C., Maynard, T., Peck, L. S., Pretty, J., Prior, S., … Watkinson, A. R. (2010). A horizon scan of global conservation issues for 2010. In Trends in Ecology and Evolution (Vol. 25, Issue 1, pp. 1–7). Elsevier Ltd. https://doi.org/10.1016/j.tree.2009.10.003

The Great Pacific Garbage Patch. (n.d.). The Ocean Cleanup. Retrieved March 7, 2020, from https://theoceancleanup.com/great-pacific-garbage-patch/?gclid=Cj0KCQiAnL7yBRD3ARIsAJp_oLZ3hf3fzBaqjwdkshNsAeVCQB2HCNvr1Qud0gbez5rnS8PaBuaW3rMaAuabEALw_wcB

Thevenon, F., Carroll, C., & Sousa, J. (2015). Plastic debris in the ocean: the characterization of marine plastics and their environmental impacts, situation analysis report. In Plastic debris in the ocean: the characterization of marine plastics and their environmental impacts, situation analysis report. https://doi.org/10.2305/iucn.ch.2014.03.en

Thomas, K., Dorey, C., & Obaidullah, F. (2019). GHOST GEAR: THE ABANDONED FISHING NETS HAUNTING OUR OCEANS Sea turtle entangled in fishing gear in the Mediterranean Sea.

Thompson, R. C., Olson, Y., Mitchell, R. P., Davis, A., Rowland, S. J., John, A. W. G., McGonigle, D., & Russell, A. E. (2004). Lost at Sea: Where Is All the Plastic? Science, 304(5672), 838. https://doi.org/10.1126/science.1094559

Thompson, J. (2019). Time is Running Out : The U.S. Landfill Capacity Crisis. Waste Business Journal, 1–7. https://nrra.net/sweep/time-is-running-out-the-u-s-landfill-capacity-crisis/

Tourinho, P. S., Ivar do Sul, J. A., & Fillmann, G. (2010). Is marine debris ingestion still a problem for the coastal marine biota of southern Brazil? Marine Pollution Bulletin, 60(3), 396–401. https://doi.org/10.1016/j.marpolbul.2009.10.013

Turner, J. (2002). Zooplankton fecal pellets, marine snow and sinking phytoplankton blooms. Aquatic Microbial Ecology, 27(1), 57–102. https://doi.org/10.3354/ame027057

Turner, J. T. (2004). The importance of small planktonic copepods and their roles in pelagic marine food webs. Zoological Studies, 43(2), 255–266.

Turner, J. T. (2015). Zooplankton fecal pellets, marine snow, phytodetritus and the ocean's biological pump. In Progress in Oceanography (Vol. 130, pp. 205–248). Elsevier Ltd. https://doi.org/10.1016/j.pocean.2014.08.005

Ugolini, A., Ungherese, G., Ciofini, M., Lapucci, A., & Camaiti, M. (2013). Microplastic debris in sandhoppers. Estuarine, Coastal and Shelf Science, 129, 19–22. https://doi.org/10.1016/j.ecss.2013.05.026

UNEP. (2014). UNEP YEAR BOOK 2014: EMERGING ISSUES IN OUR GLOBAL ENVIRONMENT | Capacity4dev. In European Union. https://europa.eu/capacity4dev/unep/documents/unep-year-book-2014-emerging-issues-our-global-environment

United States Environmental Protection Agency. (2016). Advancing sustainable materials management: 2015 Tables and Figures Assessing Trends in Material Generation, Recycling, Composting, Combustion with Energy Recovery and Landfilling in the United States. United States Environmental Protection Agency, Office of Land and Emergency Management, Washington, DC 20460, July, 22. https://doi.org/EPA530F-18-004

Unger, B., Rebolledo, E. L. B., Deaville, R., Gröne, A., IJsseldijk, L. L., Leopold, M. F., Siebert, U., Spitz, J., Wohlsein, P., & Herr, H. (2016). Large amounts of marine debris found in sperm whales stranded along the North Sea coast in early 2016. Marine Pollution Bulletin, 112(1–2), 134–141. https://doi.org/10.1016/j.marpolbul.2016.08.027

van Sebille, E. (2015). The oceans' accumulating plastic garbage. Physics Today, 68(2), 60–61. https://doi.org/10.1063/PT.3.2697

van Wezel, A., Caris, I., & Kools, S. A. E. (2016). Release of primary microplastics from consumer products to wastewater in the Netherlands. Environmental Toxicology and Chemistry, 35(7), 1627–1631. https://doi.org/10.1002/etc.3316

Verschoor, Anja; de Poorter, Leon; Droge, Rianne; Kuenen, Jeroen; de Valk, E. (2016). Emission of microplastics and potential mitigation measures (cleaning agents, paints and tyre wear).

Vroom, R. J. E., Koelmans, A. A., Besseling, E., & Halsband, C. (2017). Aging of microplastics promotes their ingestion by marine zooplankton. Environmental Pollution, 231, 987–996. https://doi.org/10.1016/j.envpol.2017.08.088

Webb, H. K., Arnott, J., Crawford, R. J., & Ivanova, E. P. (2013). Plastic degradation and its environmental implications with special reference to poly(ethylene terephthalate). Polymers, 5(1), 1–18. https://doi.org/10.3390/polym5010001

Weithmann, N., Möller, J. N., Löder, M. G. J., Piehl, S., Laforsch, C., & Freitag, R. (2018). Organic fertilizer as a vehicle for the entry of microplastic into the environment. Science Advances, 4(4), eaap8060. https://doi.org/10.1126/sciadv.aap8060

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

Wieczorek, A. M., Croot, P. L., Lombard, F., Sheahan, J. N., & Doyle, T. K. (2019). Microplastic Ingestion by Gelatinous Zooplankton May Lower Efficiency of the Biological Pump. Environmental Science and Technology, 53(9), 5387–5395. https://doi.org/10.1021/acs.est.8b07174

Wirtz, K. W. (2012). Who is eating whom? Morphology and feeding type determine the size relation between planktonic predators and their ideal prey. Marine Ecology Progress Series, 445, 1–12. https://doi.org/10.3354/meps09502

Wishner, K. F., Seibel, B. A., Roman, C., Deutsch, C., Outram, D., Shaw, C. T., Birk, M. A., Mislan, K. A. S., Adams, T. J., Moore, D., & Riley, S. (2018). Ocean deoxygenation and zooplankton: Very small oxygen differences matter. Science Advances, 4(12), eaau5180. https://doi.org/10.1126/sciadv.aau5180

WoRMS. (n.d.). Copepoda. World Register of Marine Species. Retrieved March 8, 2020, from http://www.marinespecies.org/aphia.php?p=taxdetails&id=1080

Wotton, R. S., & Malmqvist, B. (2001). Feces in Aquatic Ecosystems: Feeding animals transform organic matter into fecal pellets, which sink or are transported horizontally by currents; these fluxes relocate organic matter in aquatic ecosystems. BioScience, 51(7), 537–544. https://doi.org/10.1641/0006-3568(2001)051[0537:FIAE]2.0.CO;2

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

Zervoudaki, S., Krasakopoulou, E., Moutsopoulos, A., Protopapa, M., Gazeau, F. (2015). Effects of ocean acidification on zooplankton during a mesocosm experiment in the NW Mediterranean Sea. Proceedings 11th Panhellenic Symposium Oceanography and Fisheries, 861–864. https://www.researchgate.net/publication/316256275_Effects_of_ocean_acidification_on_zooplankton_during_a_mesocosm_experiment_in_the_NW_Mediterranean_Sea

Zhang, C., Jeong, C. B., Lee, J. S., Wang, D., & Wang, M. (2019a). Transgenerational Proteome Plasticity in Resilience of a Marine Copepod in Response to Environmentally Relevant Concentrations of Microplastics. Environmental Science and Technology, 53(14), 8426–8436. https://doi.org/10.1021/acs.est.9b02525

Zhang, P., Yan, Z., Lu, G., & Ji, Y. (2019b). Single and combined effects of microplastics and roxithromycin on Daphnia magna. Environmental Science and Pollution Research, 26(17), 17010–17020. https://doi.org/10.1007/s11356-019-05031-2