Buoyant Plastics at Sea:
Concentrations and Impacts

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Thesis Declaration

This thesis is presented as a series of three scientific papers, which is in agreement with the Postgraduate and Research Scholarship Regulation 1.3.1.33 of the University of Western Australia.


Some parts of the other chapters (chapters 1 and 5) have also been previously published in journals, magazines, and reports produced by J. Reisser during her PhD candidature. Main supervisor C. Pattiaratchi provided guidance for the entire thesis and scientific publications. The contributions of other collaborators to PhD chapters are mostly associated with research directions, assistance with data processing, and editorial input in manuscript drafts. The exceptions are the substantial contributions of G. Hallegraeff towards the identifications of the ‘epiplastic’ organisms described in chapter 3, and B. Slat towards designing the innovative sampling protocol presented in chapter 4. Besides the three scientific papers that compose the data chapters of this thesis, J. Reisser co-produced eleven additional scientific publications and five media articles during her PhD candidature. The full references to these manuscripts are provided in ‘Appendix 1 Outputs produced during this candidature’.

Julia Reisser
Abstract

Millimetre-sized plastics are a predominant type of marine debris floating at sea. These small macroscopic particles are numerically abundant in some marine environments, but little is known about their spatial distribution and environmental impacts. The goals of this thesis were to investigate how buoyant plastics are distributed in sea surface waters (both horizontally and vertically), and characterise organisms and textures on the surface of millimetre-sized marine plastics. This work is the first to (1) quantify plastic contamination levels in Australian waters, (2) characterize the biodiversity of organisms living on millimetre-sized plastics from waters around Australia, and (3) obtain high-resolution depth profiles (0 – 5 m) of plastic pollution in an oceanic accumulation zone. I collected 839 pieces of plastic in 171 surface net tows from surface waters around Australia, and 12,751 pieces of plastic in 12 multi-level tows from an oceanic accumulation zone in the North Atlantic. Plastics were mostly fragments resulting from the breakdown of larger objects (e.g. packaging and fishing gear) made of polyethylene and polypropylene polymers. Contamination levels in waters around Australia were similar to those in other marine regions (e.g. Caribbean Sea and Gulf of Maine), but considerably lower than those found in plastic pollution hotspots within subtropical gyres and Mediterranean Sea. There was a wide range of microbes and a few invertebrates on the surface of floating plastics from Australia-wide sample collections. Diatoms were particularly diverse, represented by 14 genera, 11 of which are new records of ‘epiplastic’ organisms. Plastic pollution levels in the North Atlantic accumulation zone decreased exponentially with water depth, with decay rates decreasing as wind strength increased. Plastic mass per cubic metre of water decreased more rapidly with depth than the number of plastic pieces per cubic metre, as the smaller plastic pieces were associated with lower rising velocities and were more susceptible to vertical mixing. This thesis contributed towards the global efforts of quantifying plastic contamination levels and impacts in surface waters. It highlights the widespread distribution of anthropogenic polymers, which has created a new pelagic habitat for microorganisms and invertebrates. Plastic inhabitants seem to be invading non-native marine regions by plastic transport, and playing an important role on ocean plastic degradation.

Keywords: microplastics, marine debris, plastic pollution, Australia, garbage patches
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Chapter 1  General Introduction

1.1  Plastic: definition and major types

Plastics are a diverse group of synthetic materials predominantly derived from petrochemicals, such as petroleum and natural gas. They possess a peculiar molecular architecture consisting of long chainlike macromolecules known as polymers, which are a sequence of repeating units, called monomers. Plastics can be divided into two major categories: thermoplastics and thermosets.

Thermosets are used in a few high-volume applications, such as automobile tires. They can be considered a large molecule that is destroyed with heating, meaning recycling options are mostly limited to energy recovery and physical grinding into powder (Pickering, 2006). Environmental impacts of thermosets to marine ecosystems are outside the scope of this thesis, mainly because nearly all thermosets are heavier than water. Therefore, these materials do not predominate in the top layer of the world’s oceans, which is the marine region examined in this thesis.

Thermoplastics, which will be referred here as ‘plastics’, are a group of materials made of large polymeric molecules held together by relatively weak intermolecular forces. They soften upon heating and return to their original condition when cooled. This property makes them suitable for moulding and extrusion into films, fibres and packaging. It also allows recycling into new products by re-melting and processing into new shapes. Major thermoplastic types include polyethylene (PE), polypropylene (PP), polyvinyl chloride (PVC), and polystyrene (PS). These are produced in large volumes (Figure 1.1) and are therefore of particular environmental significance.
The density of a certain plastic determines a key behaviour in the marine environment: whether the material will sink or float in seawater. All major polymers containing elements other than hydrogen and carbon are heavier than water due to their strong intermolecular forces. As a consequence, the only polymer group containing materials lighter than water are pure hydrocarbons.

A major group of pure hydrocarbons is polyolefins. They have a density range of approximately 900 – 960 kg m\(^{-3}\), thus floating in both water (1000 kg m\(^{-3}\)) and seawater (1025 – 1045 kg m\(^{-3}\)). Polyolefins are the most common type of synthetic polymer, with a share of approximately 40% of the global plastic production (Taylor, 2015). This group includes polypropylene (PP), low-density polyethylene (LDPE), linear low-density polyethylene (LLDPE), and high-density polyethylene (HDPE) materials. Polystyrene is also a pure hydrocarbon, but due to the benzene rings in its
macromolecule, it commonly sinks in water. The exception is expanded polystyrene (i.e. Styrofoam), which is highly buoyant due to its air bubbles.

Polyethylenes, an important member of the family of polyolefin resins, are the most widely used type of plastics in the world (37% of global production). They are made from the polymerization of ethylene monomers (CH₂=CH₂) and are used in products ranging from clear food wrap and shopping bags, to detergent bottles and fish crates. Polyethylenes’ durability, light-weight characteristics, and high use in single-use packaging, makes them a major type of floating pollutant that persist for long periods of time in both freshwater and marine environments, including the sea surface (Hidalgo-Ruz et al., 2012).

1.2 The Plastic Age: plastic production and waste

Due to the ideal properties of plastic for many applications (e.g. inexpensive, lightweight, flexible, durable, water resistant), it is displacing materials like paper, glass, and metal from traditional usages, and is leading to the creation of new products with high demands. Consequently, the global production of plastics has been growing exponentially since the 1950-60s, when most basic polymer groups were already available for use in a diverse range of applications (Thompson et al., 2009) (Figure 1.2). For instance, in 2012 alone, 288 million tons of plastic were produced (PlasticsEurope, 2013), which is approximately the same weight of the entire human biomass (Walpole et al., 2012). Approximately 8% of today’s annual fossil fuel production goes towards plastic production, with 4% of this converted directly into plastic materials and a similar quantity used as energy for its manufacture (Hopewell et al., 2009).
One of the key drivers of this plastic production growth is the packaging market. Approximately 37 - 44% of all plastic produced each year is used in packaging (Industry Canada, 2011, Plastic Waste Management Institute, 2013, PACIA, 2011, PlasticsEurope, 2013). This includes the manufacture of products entirely made of plastic, such as bottles, cups, bags, containers and trays, as well as multilayer structures containing plastic liners, including aluminium/tin cans and milk/juice cartons. This packaging, together with agricultural films and other disposable consumer items, represent half of the plastics produced nowadays, which are predominantly used once and disposed of in less than one year (Selke, 2003). Packaging also represents the main source of plastic waste, with 45.8% and 58% of the plastic waste produced respectively in the UK and Japan coming from packaging (Hopewell et al., 2009, Plastic Waste Management Institute, 2013).
Unfortunately, the vast majority of global plastic winds up in landfills (Hopewell et al., 2009, Hoornweg and Bhada-Tata, 2012), with only around 3.5 – 15% being recycled. In the US, 32,000,000 tonnes of plastic waste were generated in 2012, of which 9% was recycled (EPA, 2013). In Australia, 1,476,690 tonnes of plastic were used in 2011-2012, of which 20.4% was recycled (PACIA, 2011). Japan, which is a major leader in plastic waste recovery, recycled 26% of the 95,200,000 tonnes of plastic waste produced in 2011 (Plastic Waste Management Institute, 2013). Among the difficulties to increase the rates of plastic recycling are the relative high costs of processing waste into recycled materials, and the challenges to sort domestic waste into single polymer types (Hopewell et al., 2009). Apart from the small quantities of waste diverted back into the manufacture system through recycling, disposed plastic is either incinerated (with or without energy recovery) or disposed in landfills and dumps. Due to low rates of recycling and high durability, plastics are accumulating in many types of habitats worldwide, particularly in aquatic environments (Figure 1.3).
Particularly concerning issues associated with this sharp rise in plastic production and waste are (1) the toxicity of certain plastics to human health, leading to adverse effects such as increased risk for cancer and neurological problems (Breast Cancer Fundation, 2013), and (2) the impacts on marine life arising from the widespread occurrence of discarded plastics in the oceans (United Nations Environment Programme, 2014). The latter is the topic of this thesis.

1.3 Oceans: the ultimate sink for plastic pollution

Our massive plastic production and waste, the obstacles to recycle and properly dispose of plastic products, and the sharp rise in the number of ships and coastal developments, are all leading to an increase in the amount of plastic items accumulating in the oceans. Plastics can be transported from populated areas to the marine environment by rivers, wind, tides, rainwater, storm, drains, sewage disposal, and flooding; or can directly
reach the sea from vessels (e.g. fishing gear) and offshore installations (e.g. oil rigs and aquaculture farms) (Ryan et al., 2009). The relative importance of these different pathways to the total load of plastics at sea has not yet been quantified. However, some major sources are evident: (1) rivers, which can connect even inland populated areas to oceans, (2) vessels, particularly those engaged in fishing activities using plastic gear (e.g. long and heavy nets), and (3) tides, which takes litter left at beaches by users. It was recently estimated that around 4.8 to 12.7 million tonnes of plastic waste enters the oceans from land-based sources (Jambeck et al., 2015).

Once plastics reach the oceans, their fate will depend on their characteristics (e.g. density, shape and size) and the environmental conditions they are exposed to. Plastics made of resins denser than seawater (e.g. polyethylene terephthalate water bottles) will sink to the seafloor, while less dense plastics (e.g. polypropylene water bottle caps) will float at the sea surface layer for a variable period of time. Floating period will depend on processes such as hydrodynamics, debris characteristics, and ecology of the surrounding environment (Cózar et al., 2014, Eriksen et al., 2014). For instance, buoyant plastics from coastal sources may encounter strong inshore winds and currents and strand (either permanently or temporarily) in coastal environments such as sandy beaches, rocky shores, and mangroves (Carson et al., 2011). Furthermore, biofouling, which is the colonization of the debris surface by microorganisms and invertebrates, can increase the density of buoyant debris to a point where they sink to the seafloor. This process is particularly quick (days to months) for plastic items with high surface area to volume ratios (Ryan, 2015), such as plastic bags and wraps. These biofouled items may rest on the seafloor permanently and/or return to the water column, depending on the biofouling dynamics and local sediment rates (Ye and Andrady, 1991).
Throughout their marine journey, plastics slowly degrade and become brittle, then break down into progressively smaller pieces (Andrady, 2011). There are four mechanisms by which plastics degrade in the environment: photo-oxidative degradation, thermal oxidation, hydrolysis, and biodegradation (Gregory and Andrady, 2003). Common plastics encountered in marine environments (e.g. polyolefins), however, break down primarily through photo-thermal oxidation processes (i.e. mostly due to the effect of sunlight and heat). Ultraviolet light from the sun gives the energy to begin the incorporation of oxygen atoms into the polymer (Andrady, 2011), which then causes the plastic to become brittle and break into progressively smaller pieces. When the polymer chains reach sufficiently low molecular weight, microorganisms may then convert the polymer carbon into carbon dioxide or incorporate it into biomolecules (Zheng et al., 2005). The process explained above can take 50 or more years to be completed (Webb et al., 2012).

At sea, plastic degradation is particularly slow mostly due to the low temperatures and low oxygen availability. Furthermore, in the ocean the rate of hydrolysis is insignificant for most polymers. In the case of negatively buoyant plastics, degradation is likely to be even slower as ultraviolet wavelengths from the sun are readily absorbed by water, making the degradation process limited to thermal oxidation. Furthermore, biofouling on the surface of floating plastics may protect the material from exposure to sunlight, yielding a slower degradation relative to exposure on land (Gregory and Andrady, 2003).

The fragments resulting from the degradation of plastic objects are known as secondary microplastics when smaller than 5 mm. In addition to these fragments, plastics can also be directly manufactured in small sizes (< 5 mm). These are known as primary microplastics and include: pellets, the raw material used to produce plastic items (Mato
et al., 2001); synthetic fibers, including those used in clothing (Browne et al., 2011); and microbeads, which are plastic spheres found in cosmetics such as toothpaste and facial scrubs (Fendall and Sewell, 2009).

Buoyant microplastics are widespread across oceans, with well-known hotspots occurring at surface waters of the Mediterranean Sea and at large oceanic accumulation zones formed within subtropical gyres (Cózar et al., 2014, Eriksen et al., 2014). It is at the sea surface that some major and highly visible environmental impacts occur, including plastic entanglement and ingestion by pelagic animals and wide transport of fouling organisms across oceans (Barnes, 2002, Derraik, 2002, Mato et al., 2001, Wilcox et al., 2013).

The ocean’s subtropical gyres are five large continuous loops of flowing water (Talley et al., 2011b) that occupy around 40% of the Earth’s surface. Their horizontal extension is from about 10° north and south of the equator to about 45° in each hemisphere; the water circulating in these massive regions reaches nearly 2 km beneath the sea surface (Pedlosky, 1990). Those in the North Hemisphere rotate clockwise (North Pacific and North Atlantic Gyres), while those in the South Hemisphere spin counter clockwise (South Pacific, South Atlantic, and Indian Gyres). They are shaped by a strong and narrow “western boundary current” and a weak and broad “eastern boundary current”. The sea surface interiors of these gyres have low concentrations of nutrients and biomass throughout the year, but their immense size makes their total biological productivity significant to the world’s ocean ecosystem (McClain et al., 2004).

Accumulation zones of buoyant plastics within subtropical gyres can exceed 100,000 pieces km$^{-2}$ and their horizontal extensions have been inferred by numerical (Lebreton et al., 2012) and statistical modelling using satellite-tracked drifting buoys (Maximenko
et al., 2012, van Sebille et al., 2012). Buoyant plastics are transported and accumulated in these oceanic areas due to a combination of geostrophic current forcing, controlled by pressure gradients (i.e. by the level “tilt”), and the effect of local wind through (1) direct wind force applied to the surface of the floating debris - the so-called ‘windage’, (2) Stokes drift created by local waves, and (3) Ekman currents (Maximenko et al., 2012). Mean streamlines resulting from this forcing form a large-scale pattern with five well-defined convergent zones known as accumulation zones or ‘garbage patches’ (Maximenko et al., 2012) (Figure 1.4).
Figure 1.4 Top map: Mean streamlines as a combination of the mean geostrophic and Ekman velocities (Maximenko et al., 2009). Bottom map: drifter model solution after 10 years of integration from an initially homogeneous state. See details in (Maximenko et al., 2012).

Colours in the top map indicate the magnitudes of mean velocities used to compute the streamlines, and colours in the bottom map indicate the relative concentration of drifters/debris. Map sources: Maximenko et al. 2009 and Maximenko et al. 2012.

The main advantage of the statistical modelling approach developed by Maximenko et al. 2012 is its ability to describe motions of floating objects, even without a full understanding of the tremendously complex dynamics of the upper ocean. Their model indicates that tropical and subpolar regions are cleared from buoyant debris within three
years, with most of this pollution being transported into the five subtropical gyres. In ten years time, it predicts that floating debris are redistributed within and between the subtropical gyres to form more compact accumulation zones, centred at around 30 degrees latitude. The North and South Pacific accumulation zones are located in eastern parts of the corresponding subtropical gyres, while the North and South Atlantic as well as the South Indian accumulation zones are elongated zonally across their entire ocean basin (see Figure 1.4).

1.4 Marine plastic pollution impacts

Animals as small as microscopic zooplankton (Wright et al., 2013) and as large as whales (Fossi et al., 2012) ingest plastic debris. Plastics can also enter animals’ bodies through respiration, as has been reported in crabs (Watts et al., 2014). Ingestion of plastic items by marine animals can lead to gastrointestinal perforation and blockage, reduction of food intake, reproductive disorders, and death (Derraik, 2002). At least 170 marine species are affected by plastic ingestion, including threatened species of seabirds, turtles and mammals (Vegter et al., 2014).

Chemical impacts of plastics on organisms, food webs, and ecosystems have also become a focus of concern over the last decade, e.g. (United Nations Environment Programme, 2014). One of the main reasons for such concern is that over half of our plastic objects contain at least one ingredient classified as hazardous (Rochman et al., 2013a). Many plastic products contain non-polymeric components (e.g. residual monomers, oligomers, low molecular weight fragments, catalyst remnants, polymerisation solvents and additives) that can be carcinogenic, mutagenic, and/or toxic for organisms, with potential long-term effects (Lithner, 2011). Since these substances are usually of low molecular weight and are weakly/not bound to the polymeric
macromolecules, they and/or their degradation products can be released from the plastic into air, water or other contact media, such as food (Lithner, 2011).

Furthermore, plastics that enter aquatic environments can become increasingly hazardous by adsorbing persistent organic pollutants and metals on their surface (Rios et al., 2007, Holmes et al., 2012). Due to the hydrophobic nature of plastics, hydrophobic pollutants such as polychlorinated biphenyls - PCBs (Mato et al., 2001) and polycyclic aromatic hydrocarbons - PHAs (Rios et al., 2007) accumulate on their surfaces. Adsorption of trace metals on plastic debris can also occur through exposure to environmental conditions that lead to the development of viable surface sites by photo-oxidation, biofouling and deposition of sediment particles (Holmes et al., 2012). When plastic enters the body via ingestion or other means, these concentrated pollutants can be transferred to predators and also up their food chains. Such a bio-magnification process is more likely to occur when plastics are small enough to be ingested by low trophic fauna, such as small fish and zooplankton (Browne et al., 2013, Rochman et al., 2013c).

Entanglement in plastic items, especially discarded fishing gear (e.g. ropes, straps, lines, pots, traps, nets), are also a serious threat to some species of marine vertebrates (Derraik, 2002). It can lead to physical injuries, drowning, increased drag, impairment of abilities to forage and avoid predators, and ultimately death. At least 135 marine species have been recorded entangled in marine debris (Vegter et al., 2014). Animals that often occur at the sea surface, such as air-breathing vertebrates (e.g. sea turtles, birds, and mammals), are particularly prone to this type of adverse interaction. Young fur seals, which are both curious and playful, are often entangled in nets and packing bands (Derraik, 2002). Entanglement of juvenile northern sea lions (Eumetopias jubatus), Hawaiian monk seals (Monachus schauinslandi), and northern fur seals
(Callorhinus ursinus) has been listed as one of the factors contributing to the decline of their populations (Derraik, 2002). Benthic organisms, especially those with branching morphologies (e.g. gorgonians, sponges, corals), are also affected by tissue abrasion and mortality caused by entanglement in lost fishing gear (Chiappone et al., 2005).

On the other hand, many marine species can also benefit from the occurrence of plastics at sea, which is a new long-lasting type of floating habitat. These include fish species and fouling organisms that can invade non-native waters through plastic drifting (Barnes, 2002) and ‘epiplastic’ pathogens, which may infect animals that ingest plastics (Pham et al., 2012). We still know very little about the dwellers of the widely dispersed and abundant microplastics. The environmental implications of the occurrence of organisms on the surface of millimetre-sized plastics are discussed in Chapter 3, where the inhabitants of millimetre-sized plastics from waters around Australia are described.

Another environmental impact of plastic contamination is the alteration of physical properties of marine environments. It changes light, oxygen and refuge availability, as well as heat transfer and water movement throughout sediments (Goldberg, 1997, Carson et al., 2011). Such effects could potentially have consequences to benthic communities and animals that have offspring sex determined by sand temperature, such as sea turtles (Goldberg, 1997, Carson et al., 2011). Plastic-induced alterations to natural environments also lead to social and economic impacts (Vegter et al., 2014), such as decreased tourism in beaches and diving destinations heavily polluted by marine litter.

1.5 Monitoring plastic pollution

To gain a better understanding of the environmental hazards associated with marine plastic pollution, several studies have attempted to quantify marine plastic debris,
ranging from several meters to 0.001 millimetres in size (Hidalgo-Ruz et al., 2012, Pichel et al., 2012). Such investigations have sampled plastics from the shoreline, seafloor, water column, and sea surface (Thompson et al., 2004, Law et al., 2010, Browne et al., 2011, Kukulka et al., 2012, Schlining et al., 2013, Van Cauwenberghe et al., 2013b).

Floating marine plastics have been quantified mostly through visual counts conducted from vessels and airplanes (Pichel et al., 2007, Hinojosa et al., 2011, Pichel et al., 2012), and by sampling devices that collect plastics from the oceans. Such instruments include Continuous Plankton Recorders (Thompson et al., 2004), Niskin bottles (Gordon, 2000), rotating drum samplers (Ng and Obbard, 2006), and zooplankton nets (Carpenter and Smith, 1972).

1.5.1 Visual surveys

The most common method of estimating amounts of large plastics in surface waters is by counting floating objects while aboard vessels. Generally, an observer stands on the flying bridge looking for floating debris as the ship moves through the area. Binoculars are sometimes used to confirm the characteristics of the sighted objects (e.g. material, size, colour). The area that is visually scanned for floating debris varies between studies. Surveys following the Strip Transect approach stipulate a maximum distance from the vessel in which the observer should scan for debris. It is a very simple method that assumes all objects within the scanned area are counted (100% probability detection). Surveys following the Line Transect approach (Buckland et al., 2005), have the observers focusing their search effort in the heading line of the vessel, and use the perpendicular distances between the sighted objects and the vessel’s heading line to estimate detection probability functions, e.g. (Titmus and Hyrenbach, 2011). To obtain
an estimate of abundance, the number of objects observed during a certain time is divided by the sampled area, which is equal to the distance travelled by the vessel (transect length) multiplied by the distance from the boat where plastics were counted (transect width). These plastic concentrations are commonly reported in number of items per area, but grams per area are also estimated in a few studies that used weights of beached plastic items to infer the mass of sighted objects and transform numerical densities into mass densities (Eriksen et al., 2014).

Plastic concentrations in pieces km\(^{-2}\) reported in 18 studies that conducted visual surveys are plotted in the top panel of Figure 1.5 (Venrick et al., 1973, Morris, 1980a, Dixon and Dixon, 1983, Dahlberg and Day, 1985, Day and Shaw, 1987, Ryan, 1990, Dufault and Whitehead, 1994, Aliani et al., 2003, Thiel et al., 2003, Barnes and Milner, 2005, Shiomoto and Kameda, 2005, Hinojosa and Thiel, 2009, Titmus and Hyrenbach, 2011, Zhou et al., 2011, Williams et al., 2011, Ryan, 2013b, Ryan, 2013a, Thiel et al., 2013). There are many variables that influence these reported estimates, including sea state, distance from which objects were observed, minimum plastic size counted, etc. As such, comparisons between studies should be done with caution.
Generally, centimetre-sized fragments resulting from the disintegration of larger plastic objects were the most common type of debris observed during these visual surveys, particularly in offshore regions (Venrick et al., 1973, Dahlberg and Day, 1985, Titmus and Hyrenbach, 2011). Entire plastic items, such as bags, Styrofoam blocks, bottles,
packaging, and fishing gear, were also commonly sighted, especially in coastal waters (Thiel et al., 2003, Williams et al., 2011).

Sighted plastics were widespread in the sampled marine regions and mean concentrations higher than 10 pieces km\(^{-2}\) occurred close to coastal populated areas (i.e. Indonesian, Chilean, Canadian, and South African waters) as well as in oceanic accumulation zones within subtropical waters of the North Pacific and in the Mediterranean Sea.

There are still vast areas of the oceans to be sampled, and more data is required to adequately document the location of large-scale concentrations of plastic objects. During my PhD, I did a few visual surveys in waters around Australia, but the results are not presented in this thesis. They were used in a recent global meta-analysis estimating the distribution and load of plastics at the world’s sea surface (Eriksen et al., 2014). It was published while this thesis was under review.

### 1.5.2 Surface net tows

Zooplankton nets, such as Neuston and Manta nets (Brown and Cheng, 1981) are by far the most common devices used to sample small pelagic plastics (Hidalgo-Ruz et al., 2012). They are towed from vessels to systematically sample buoyant plastics at the air-seawater interface, where floating material tend to accumulate (Kukulka et al., 2012). In comparison to the Neuston net, the Manta net requires more people and logistics to be deployed. On the other hand, it has two important advantages in relation to the Neuston net: (1) its paravanes steer the net at an angle to the ship’s path, thus avoiding the vessel’s wake influence in the sampling, and (2) the top edge of the net always rides on the surface, ensuring a constant sampling of the sea surface and an easy calculation of the area and volume sampled by each net tow (Brown and Cheng, 1981).
The main advantage of these surface nets is their capacity to concentrate buoyant material from a relatively large volume of water (Hidalgo-Ruz et al., 2012). After each net tow, the content captured by the net is carefully examined to separate plastics from biological material. Detected plastics are then counted and/or weighed and usually reported in pieces per area (Hidalgo-Ruz et al., 2012), although pieces per volume, mass per area, and mass per volume are also used.

The findings of 15 studies that conducted surface net tows are shown in the lower panel of Figure 1.5 (Carpenter and Smith, 1972, Shaw, 1977, Morris, 1980b, Wilber, 1987, Ryan, 1988, Gregory, 1990, Day et al., 1990, Dufault and Whitehead, 1994, Moore et al., 2001, Yamashita and Tanimura, 2007, Law et al., 2010, Zhou et al., 2011, Collignon et al., 2012, Eriksen et al., 2013, Van Cauwenbergher et al., 2013a). These studies report plastic pollution levels in pieces km$^{-2}$. There are many variables that influence these reported estimates, including sampling design, net mesh size, and processing technique used, as well as efforts towards finding and identifying plastic particles smaller than 1 mm. As such, comparisons between studies should be done with caution.

Fragmented pieces of larger plastic objects were by far the most common plastic type described in the “net tow” reports considered here. Plastic pellets had a high relative abundance in reports from the 1970-80s, but decreased thereafter (Law et al., 2010). This is probably due to both an increase in amounts of secondary microplastics, and a decrease in pellets being lost during transportation.

Small marine plastics, mostly less than 5 mm across (microplastics), were widespread in the sampled marine regions, and mean concentrations higher than 10,000 pieces km$^{-2}$ were present in the Mediterranean Sea and in oceanic subtropical areas of the North Pacific, South Pacific, and North Atlantic. By considering the estimates from these
plastic pollution observations, some recent global studies (Cózar et al., 2014, Eriksen et al., 2014), and the outputs of global models of plastic dispersal (Lebreton et al., 2012, Maximenko et al., 2012, van Sebille et al., 2012), it is possible to confidently conclude that small plastics are concentrated within all large subtropical areas of the oceans (“oceanic gyres”), as well as in the Mediterranean Sea. There are still several gaps in the global datasets, particularly in the southern hemisphere and high latitudes. As such, other large-scale accumulation zones may exist. For instance, the model developed by Van Sebille et al. (2012) predicted an extra accumulation zone in the Barents Sea, while the Lebreton et al. (2012) model identified many coastal accumulation zones, most of which are still unsampled.

### 1.5.3 Subsurface net tows and depth profile modelling

Most of what is known about at-sea buoyant plastic characteristics and concentrations comes from surface net sampling. However, buoyant plastics can be transported to deeper waters due to vertical water movements created by turbulence and other circulation patterns, such as Langmuir circulation (Kukulka et al., 2012).

Lattin et al. (2004) performed paired manta (surface) and bongo (5 m) net tows in Santa Monica Bay before and shortly after a storm event, suggesting that high wind conditions and urban runoff enhance vertical mixing of plastic debris, both at the sea surface and ocean floor. Doyle et al. (2011) collected surface and subsurface (15 m from the bottom) samples during four cruises off the US west coast. They found higher quantities of plastic at the sea surface, with the occurrence of subsurface plastics only during a winter cruise. They also attributed the presence of plastics in the water column to the mixing of particles from surface and sediments.
Kukulka et al. (2012) conducted the first comprehensive multi-level survey of buoyant plastics. These authors used a Neuston net and a multiple-net Tucker Trawl (Hopkins et al., 1973) to sample plastics at the surface, 5 m, 10 m, and 20 m deep. Their observations were used to validate a model capable of predicting depth-integrated plastic numerical concentrations (pieces km\(^{-2}\)) using surface values and wind conditions.

Their model assumes that buoyant plastics are vertically distributed due to wind-driven mixing and that the depth-integrated plastic concentrations \(C_i\), pieces km\(^{-2}\) can be inferred by applying the following one-dimensional column model:

\[
C_i = \frac{C_s}{1 - e^{-dwbA_o}}
\]

Where: \(C_s\) = surface plastic concentration (pieces km\(^{-2}\)); \(d\) = immersion depth of the surface-towed net; \(w_b\) = buoyant rise velocity of marine plastics; \(A_o\) = near-surface turbulent (eddy) exchange coefficient, which was estimated by the following formula:

\[
A_o = 1.5u_{*w}kH_s
\]

Where: \(k\) = von Karman constant (equal to 0.4); \(H_s\) = significant wave height (m); \(u_{*w}\) = frictional velocity of water (m s\(^{-1}\)).

According to this model, one of the main drivers of the vertical distribution of plastic pollution is the rising velocity of plastic particles (Figure 1.6). Preliminary experiments conducted in Kukulka et al. 2012 indicated that this velocity ranges from 0.007 – 0.014 m s\(^{-1}\). However, details of this experimental work were not provided in the publication. Furthermore, their comparison between observed depth profiles and model estimates was incomplete because their model predicted an exponential depth decay on plastic
pollution levels, with the largest decrease in the upper two metres, where no subsurface measurements were taken (Kukulka et al., 2012) (see Figure 1.6).

Figure 1.6 Observed (dots) and modelled (lines) depth profiles of normalized plastic numerical concentration for different values of frictional velocity of water ($u^*$) and debris rising speed ($w_b$). Note the strong dependence between $w_b$ and the depth profile of plastic concentration. Source: (Kukulka et al., 2012)

A better understanding of the vertical transport of buoyant plastics is fundamental for improving estimates of ocean plastic load, size distribution, and dispersal (Kukulka et al., 2012, Law et al., 2014, Isobe et al., 2014). In this context, Chapter 4 of this thesis investigates the wind-driven turbulent transport of buoyant plastics by collecting high-resolution observations of plastic concentrations and characteristics (e.g. rising speeds) in the ocean top layer (0 – 5 m).
1.6 Goals and aims

Despite growing public awareness of ocean plastic pollution, the abundance, spatial
distribution, and ecological implications of marine plastic debris are still poorly
evaluated. It is clear that plastic pollution is a serious environmental issue occurring in
all oceans, and deserves further research.

The goals of this thesis were to investigate how buoyant plastics are distributed in sea
surface waters (both horizontally and vertically), and characterise organisms on the
surface of millimetre-sized marine plastics.

I collected plastic samples from surface waters around Australia (N = 171 15-minute
surface net tows, 839 pieces), and from an oceanic plastic pollution hotspot (N = 12 1-
hour net tows, 12,751 pieces) to achieve the following aims:

Aim 1: Quantify plastic contamination levels in waters around Australia.

Aim 2: Describe textures and organisms on the surface of millimetre-sized plastics from
Australian waters.

Aim 3: Investigate the vertical profile of plastic pollution in the top layer (0 – 5 m) of
the oceans.

1.7 Structure of the thesis

This thesis was prepared as a ‘series of papers’, following the guidelines of the
University of Western Australia. It has five chapters: this general introduction chapter,
three data chapters, and a general discussion chapter.
This chapter covered marine plastic pollution background in order to justify the overall goals of the thesis. Additional literature reviews, focused on the aims and specific objectives, are presented in the introduction sections of each data chapter.

The bodies of work related to aims 1, 2 and 3 are described in chapters 2, 3, and 4, respectively. These chapters were written in standard scientific publication format, so they can be read individually or as a part of the whole thesis. These three manuscripts (two published in PLOS ONE and one in Biogeosciences) are generally reproduced verbatim, except for:

- Acknowledgements, which have been consolidated into one general thesis acknowledgement section;
- References, which have been consolidated into a general thesis references section, following the Harvard bibliography style;
- Table and figure numbers, which are now following the thesis structure;
- Citations to the papers arising from this thesis, which have been changed to references to the corresponding thesis chapters;
- Language, which has been changed from American to British English.

Finally, chapter 5 brings together the main findings of this thesis, highlighting their scientific significance and limitations, and suggests directions for future research related to marine plastic pollution.
Chapter 2  Marine plastic pollution in waters around Australia: characteristics, concentrations, and pathways

2.1 Summary

Plastics represent the vast majority of human-made debris present in the oceans. However, their characteristics, accumulation zones, and transport pathways remain poorly assessed. We characterised and estimated the concentration of marine plastics in waters around Australia using surface net tows, and inferred their potential pathways using particle-tracking models and real drifter trajectories. The 839 marine plastics recorded were predominantly small fragments (“microplastics”, median length = 2.8 mm, mean length = 4.9 mm) resulting from the breakdown of larger objects made of polyethylene and polypropylene (e.g. packaging and fishing items). Mean sea surface plastic concentration was 4256.4 pieces km\(^{-2}\), and after incorporating the effect of vertical wind mixing, this value increased to 8966.3 pieces km\(^{-2}\). These plastics appear to be associated with a wide range of ocean currents that connect the sampled sites to their international and domestic sources, including populated areas of Australia’s east coast. This study shows that plastic contamination levels in surface waters of Australia are similar to those in the Caribbean Sea and Gulf of Maine, but considerably lower than those found in the subtropical gyres and Mediterranean Sea. Microplastics such as the ones described here have the potential to affect organisms ranging from megafauna to small fish and zooplankton.
2.2 Introduction

Plastics are a diverse group of materials derived from petrochemicals (Thompson et al., 2009). Their global production has grown exponentially from 1,700,000 tonnes in 1950 to 280,000,000 tonnes in 2011 (PlasticsEurope, 2012). The disposability of plastics, together with their low recycling rates, has contributed to a significant rise in the amount of waste produced globally (Hoornweg and Bhada-Tata, 2012). For instance, in Australia, 1,433,046 tonnes of plastics were used in 2010-2011, of which only 20% was recycled. Moreover, around 37% of this plastic was for the manufacturing of single-use disposable packaging (PACIA, 2011). Plastics are transported from populated areas to the marine environment by rivers, wind, tides, rainwater, storm drains, sewage disposal, and even flood events. It can also reach the sea from vessels (e.g. fishing gear) and offshore installations (Ryan et al., 2009). Once in the oceans, they will either float at the ocean surface, or sink to the seafloor if made from polymers denser than seawater (Andrady, 2011). Buoyant plastics may be cast ashore by inshore currents or winds (Thiel et al., 2013), or may enter the open ocean, where they tend to accumulate in convergence zones such as the ones formed by the five large-scale gyres: South and North Pacific, South and North Atlantic, and Indian (Moore et al., 2001, Law et al., 2010, Eriksen et al., 2013).

Marine plastics are known to undergo fragmentation into increasingly smaller pieces by photochemical, mechanical and biological processes (Andrady, 2011, Davidson, 2012). Plastics are also directly manufactured in small sizes (< 5 mm), which may find their way into the oceans. These include virgin plastic pellets (pelletwatch.org) (Mato et al., 2001), synthetic fibres from clothes (Browne et al., 2011), microbeads from cosmetics (Fendall and Sewell, 2009), and synthetic ‘sandblasting’ media (Andrady, 2011). There is increasing awareness that these small plastic particles (often called microplastics
when smaller than 5 mm) (Andrady, 2011) represent a significant proportion of the human-made debris present in the oceans. However, their at-sea spatial and temporal dynamics remain poorly assessed, mostly due to a lack of data on their characteristics and at-sea occurrence (Kukulka et al., 2012, Lebreton et al., 2012). In Australia, the only published information on microplastics comes from a global study that recorded their occurrence in the sediments of Busselton beach (Western Australia) and Port Douglas (Queensland) (Browne et al., 2011). Apart from this, our current knowledge on plastic contamination in the Australian marine environment is restricted to (1) beach litter clean-ups that record mainly the occurrence of relatively large objects, e.g. (Jones, 1995, Frost and Cullen, 1997, Edyvane et al., 2004); (2) land-based surveys of marine megafauna impacted by marine debris, e.g. (Jones, 1995, Carey, 2011, Schuyler et al., 2012, Verlis et al., 2013); and (3) inferences based on plastic pollution reports from New Zealand, e.g. (Gregory, 2009).

The impacts of plastics on marine vertebrates, such as turtles, mammals and birds, have been well recognized since the 80s (Carr, 1987, de Stephanis et al., 2013). However, only recently has concern about the effects of small plastic particles on food webs and marine ecosystems been raised. More than half of modern plastics contain at least one hazardous ingredient (Rochman et al., 2013a) and those that end up in aquatic systems can become increasingly toxic by adsorbing persistent organic pollutants on their surface (Rochman et al., 2013b). These concentrated toxins might then be delivered to animals via plastic ingestion and/or endocytosis (Teuten et al., 2009, von Moos et al., 2012) and transferred up their food webs (Basheer et al., 2004, Choy and Drazen, 2013, Gassel et al., 2013). This bio-magnification process is more likely to happen when plastics are small enough to be ingested by organisms that are close to the bottom of the ocean food web, such as planktivorous fish (Boerger et al., 2010) and zooplankton
(Cole et al., 2013). For instance, it was inferred that small plastic particles found in the stomach contents of Southern Bluefin tuna captured close to Tasmania (Young et al., 1997) were coming from the guts of their prey: myctophid fish (Eriksson and Burton, 2003). In this scenario, plastic contaminants can be transferred to the affected organism and then biomagnified up the food chain. If this process is taking place, plastics can affect the health of food webs, which include humans as an apex predator.

Australia’s acknowledgement of plastic threats to marine ecosystems is mostly limited to impacts from relatively large debris (e.g. abandoned fishing nets, plastic bags) on marine megafauna (e.g. turtles, mammals, birds) (Commonwealth of Australia, 2009). A first step towards a better understanding of the extent of marine plastic hazards to Australian organisms and environments is a better assessment of the occurrence and characteristics of plastic debris at-sea. To this end, we characterized (size, type, color, polymer) and estimated concentration (pieces km\(^{-2}\)) of plastics in waters around Australia using surface net tows. Additionally, potential pathways taken by the collected plastics were inferred using outputs of a dispersal model and trajectories of satellite-tracked drifting buoys.

### 2.3 Materials and Methods

Ethics Statement: Permits to conduct this field research were obtained from the Great Barrier Reef Marine Park Authority (GBRMPA: permit G11/34378.1). No other special permitting was required because sampling was limited to the collection of marine debris.

During seven transit voyages aboard Australian vessels (Figure 2.1), we undertook three consecutive 15-minute net tows (mean ± standard deviation tow length = 1.3 ± 0.50 km) at 57 locations (hereafter called “net stations”), while the ship was travelling at a speed
of 2 – 4 knots. These net tows sampled the air-sea interface, using a Neuston net (1.2 × 0.6 m mouth, 335 μm mesh) or a Manta net (1 × 0.17 m mouth, 333 μm mesh). After each net tow, the collected material was transferred to a container filled with seawater and examined for floating plastic pieces for at least an hour by a trained observer (J.R.). Each plastic piece was picked up with forceps and placed in a graduated dish to be counted, measured (length), photographed and classified into type (hard, soft, line, expanded polystyrene, pellet), and colour. A random sample of 200 plastic pieces was selected for polymer composition analysis by Fourier transform infrared spectrometry (FT-IR; range = 500 - 4000 cm⁻¹). Polymer type was determined by comparing sample FT-IR spectra against known spectra from a database (Perkin-Elmer ATR of Polymers Library).

![Figure 2.1 Location of the 57 net stations sampled during this study.](image)

Dot colours indicate the voyage when the net station was sampled and numbers follow the chronological order of sampling. Pictures of the two types of net used are shown in the right panel.

To estimate sea surface plastic concentrations \(C_s \text{ pieces km}^{-2}\), we first divided the number of plastic pieces found in the cod-end of each net tow by its towed area, which
was estimated by multiplying net mouth width by tow length (determined from GPS position data). Mean $C_s$ was then estimated for each of the 57 net stations by averaging the $C_s$ of its three net tows. To our knowledge, this is the first study to take net tow replicates for marine plastic sampling. Apart from providing us measurements of $C_s$ variability, our approach (i.e. execution of 3 short net tows instead of 1 long trawl) also avoided net clogging by gelatinous zooplankton.

Since buoyant plastics are vertically distributed due to wind-driven mixing, we also estimated depth-integrated plastic concentrations ($C_i$, pieces km$^{-2}$) by applying a one-dimensional column model (Kukulka et al., 2012):

$$C_i = \frac{C_s}{1 - e^{-d w_b A_o}}$$

Where:

$d$ = immersion depth of the surface-towed net; equal to 0.17 m for the Manta net tows (full immersion of the net frame) and 0.3 m for the Neuston net tows (half of the frame immersed).

$w_b$ = buoyant rise velocity of marine plastics; equal to 0.02 m s$^{-1}$. Preliminary experiments indicate that it ranges from 0.005 – 0.035 m s$^{-1}$ (Kukulka et al., 2012).

$A_o$ = near-surface turbulent (eddy) exchange coefficient, which was estimated by:

$$A_o = 1.5 \nu_w k H_s$$

Where:
\[ k = \text{von Karman constant; equal to 0.4.} \]

\[ H_s = \text{significant wave height (m).} \]

\[ u_w = \text{frictional velocity of water (m s}^{-1}). \]

Both \( H_s \) and \( u_w \) were taken from the ERA-Interim model (Dee et al., 2011). There was a considerable similarity between wind fields of the ERA-Interim forecast model \( (U_{10}) \) and the wind speed measured by an anemometer \( (w) \) on five of our seven voyages \( (U_{10} = 0.85 + 1.04w, \ r^2 = 0.79, \ N = 39 \text{ net stations}) \), indicating that the use of the model outputs is adequate.

To infer potential pathways taken by the collected plastics, we used two approaches: (1) application of the Australian Connectivity Interface Connie2 (csiro.au/connie2), and (2) trajectories of satellite-tracked buoys from the Global Drifter Program (aoml.noaa.gov/phod/dac). In our first approach, an area of 0.1° latitude by 0.1° longitude was created around each net station and particle-tracking models were run backwards in time. Particles were released within these areas over a 30-day period (25 particles per day), and subsequently tracked for a dispersal time equal to 45 days. These models were forced by averaged ocean current fields (2002 - 2006) of the month when the net station was sampled. Details of the particle tracking model, and the eddy-resolving/data-assimilating ocean general circulation model can be found in (Condie et al., 2005) and (Schiller et al., 2008), respectively. In our second approach, an area of 4° latitude by 4° longitude was centred on each net station and drifters (drogued and undrogued) that reached these regions were selected. The tracks starting from the drifter release point until they entered one of the net station areas were then plotted onto maps.
2.4 Results

We recorded 839 pieces of plastic, ranging in length from 0.4 to 82.6 mm (median = 2.8 mm, mean ± standard error = 4.9 ± 0.27 mm, Figure 2.2). The majority of these plastic pieces had low circularity in their shape when compared to manufactured plastic particles (e.g. pellets and microbeads from cosmetics), suggesting they mostly resulted from the breakdown of larger items. The main plastic type was hard plastic (N = 633, median length = 2.4 mm, range = 0.7 - 57.0 mm) followed by soft plastic (N = 142, median length = 5.0 mm, range = 0.5 - 73.0 mm), plastic line (N = 54, median length = 10.3 mm, range = 2.0 - 82.6 mm), expanded polystyrene (N = 8, median length = 2.9 mm, range = 1.3 - 24.3 mm), and pellet (N = 2, both 4 mm). Most plastics were white/transparent (84.7 %), but blue (8.3 %) and other colours (7 %) were also present. Of the 200 pieces subjected to FT-IR, 67.5 % were made of polyethylene, 31 % of polypropylene, 1 % of expanded polystyrene, and 0.5 % of ethylene vinyl acetate (Figure 2.3).
Figure 2.2 Size and types of marine plastics collected around Australia.

Bars indicate the number of plastic pieces within each size category (< 2.5, 2.5 - 4.9, 5 – 10, > 10 mm) and colours show the amount of each plastic type within size categories. Examples of the types of plastic we collected are shown in the photos, including our biggest fragment of hard plastic (length = 57 mm, net station 32), soft plastic (length = 73 mm, net station 57, note the Indonesian words), and expanded polystyrene (Styrofoam cup fragment, length = 24.3 mm, net station 28).
Approximately 80% of our net tows (136 out of 171), and 93% of our net stations (53 out of 57), had at least one piece of plastic (range: 0 – 68, median = 2, mean ± standard error = 4.9 ± 0.63 pieces per net tow). Estimated sea surface plastic concentrations ($Cs$) for each net tow ranged from 0 to 48895.6 pieces km$^{-2}$ (median = 1932.1 pieces km$^{-2}$, mean ± standard error = 4256.4 ± 757.79 pieces km$^{-2}$) and the mean $Cs$ of net stations varied between 0 and 23610.7 pieces km$^{-2}$ (Figure 2.4, Appendix 2).
Figure 2.4 Mean sea surface plastic concentration ($C_s$) at the 57 net stations.
White crosses indicate location of major Australian cities (population > 1 million). From west to east: Perth, Adelaide, Melbourne, Sydney, and Brisbane.

Relatively high mean $C_s$ (> 15500 pieces km$^{-2}$) were estimated only at low wind speeds (< 7 m s$^{-1}$, Figure 2.5a). There was an inverse relationship between $C_s$ and wind forcing ($b = -0.77$ in $C_s = a(\mu_w)b$), which was relatively consistent with the biophysical model applied here (Figure 2.5b). When taking into account the effect of wind-mixing, net tow plastic concentrations increased by a mean factor of 2.8 (range: 1.04 – 10.0, median = 1.9). Hence, the amount of plastics collected by our net tows ($C_s$) represents anywhere between 10.0 % and 96.1 % (median = 52.7 %, mean ± standard deviation = 50.0 ± 24.47 %) of the estimated total amount of plastic present in the water column ($C_i$, Figure 2.6).
Figure 2.5 Sea surface plastic concentration (Cs) versus a) wind speed (U10) and b) water friction velocity ($u^*$).

In (b) we also show the linear fit ($Cs = a (u^*)^b$) and theoretical model estimates for $Cs$, when depth-integrated plastic concentration ($C_i$) is equal to 8966 (mean $C_i$ of the 171 net tows) and significant wave height ($H_s$) is equal to the mean (1.85 m), maximum (4.78 m) and minimum (0.47 m) values estimated for the 57 net stations.
Figure 2.6 Mean and standard error of sea surface (Cs) and depth-integrated (Ci) plastic concentrations.

Blue represents mean and standard error of Cs and red represents mean and standard error of Ci.

Depth-integrated plastic concentration estimates (Ci) for each net tow ranged from 0 to 105438.6 pieces km\(^{-2}\) (median = 4363.7 pieces km\(^{-2}\), mean ± standard error = 8966.3 ± 1330.75 pieces km\(^{-2}\)) and the mean Ci of net stations ranged from 0 to 43194.5 pieces km\(^{-2}\) (Figure 2.7). In this scenario, plastic concentrations higher than 15500 pieces km\(^{-2}\) (red dots) were quite common, and those higher than 31500 pieces km\(^{-2}\) (dark red dots) were found close to populated areas (Brisbane and Fiji) as well as in some remote coastal regions (southwest Tasmania) and oceanic areas (Figure 2.7).
Figure 2.7 Mean depth-integrated plastic concentration (Ci) at the 57 net stations.
White crosses indicate location of major Australian cities (population > 1 million). From west to east: Perth, Adelaide, Melbourne, Sydney, and Brisbane.

A wide range of pathways was taken by the virtual particles arriving at the net stations (Figure 2.8, Appendix 2). The routes taken by real drifters, from their release points to the net stations, showed similar patterns but covered larger areas due to their longer drifting time and wider range of release date (Figure 2.9, Appendix 2).
Figure 2.8 Cumulative probability distribution of virtual particles arriving at the 57 net stations.  
The month when the virtual particles (25 per day) were released is indicated in each panel.
Backtracking dispersal time was equal to 45 days and arriving destinations (net stations) are
marked with purple dots. See also Appendix 2.
Figure 2.9 Real drifter pathways arriving at the 57 net stations.
Purple dots indicate net station locations and asterisks indicate drifter release areas. See also Appendix 2.
2.5 Discussion

We found that the surface waters around Australia are contaminated with small plastics that are mostly a by-product of the degradation of larger objects made of polyethylene and polypropylene. The high prevalence of plastic fragments smaller than 5 mm in Australian waters is consistent with other regions of the world’s oceans, where microplastics were found to be the most abundant type of debris in all types of marine environment (Moore et al., 2001, Thompson et al., 2004, Browne et al., 2010, Law et al., 2010, Browne et al., 2011, Eriksen et al., 2013). Plastic pollution levels were moderate when compared to concentrations in other marine areas (Moore et al., 2001, Yamashita and Tanimura, 2007, Law et al., 2010, Collignon et al., 2012, Eriksen et al., 2013). Higher amounts of plastic were found close to cities on Australia’s east coast, as well as in remote locations (west Tasmania and North West Shelf). Recent studies reported toxicological effects of these small and contaminated plastics on a host of organisms, including large marine vertebrates (Fossi et al., 2012) and fish (Basheer et al., 2004, Choy and Drazen, 2013, Gassel et al., 2013, Wright et al., 2013). As such, small plastics are a type of harmful marine debris, implying that plastic hazards to Australian species and ecological communities are likely to be broader than those officially recognized.

2.5.1 Characteristics of marine plastics

Captured plastic particles ranged in size from 0.4 - 82.6 mm. The frequency distribution of different sized plastics, which was skewed towards smaller particles, provides evidence for the existence of smaller plastics. Current methods for assessing plastic pollution at the ocean surface rely on the use of nets, which omits plastic particles...
outside the collectible range of their mesh (Hidalgo-Ruz et al., 2012). It will be critical for future investigations to develop efficient and reproducible techniques capable of detecting smaller buoyant plastic particles (micro and nanoparticles). In addition, post processing techniques for sorting particles are also likely to miss small fragments (Hidalgo-Ruz et al., 2012). An example of a new method with the potential to eliminate this limitation is the application of molecular mapping by reflectance micro-FT-IR spectroscopy, which does not rely on visual selection of plastic particles for characterization (Harrison et al., 2012).

Hard plastics were by far the most common plastic type found (75.4%), but soft plastics (e.g. fragments of plastic wrappers) and lines (mostly fishing lines) were also relatively common (16.5% and 6.4%, respectively). It is interesting to note that soft plastics were more abundant in the larger size class (> 2.4 mm). Our findings are consistent with recent studies documenting plastic pollution at the ocean surface, although explanations for variations in hard/soft plastic trends are not given (Moore et al., 2001, Eriksen et al., 2013, Morét-Ferguson et al., 2010). Plastics gradually lose buoyancy in seawater as a result of biofilm formation (Ye and Andrady, 1991). We suggest that negative buoyancy due to biofouling occurs more quickly in soft/thin than in hard/thicker plastic fragments, resulting in a decline in the occurrence of soft plastics at the ocean surface, as they become smaller/older and begin to sink. Indirect evidence for this is that the proportion of soft plastics found in our coastal net stations was higher than that reported in open ocean settings further away from potential sources (Morét-Ferguson et al., 2010). While a small number of experimental studies have confirmed that biofilms decrease the buoyancy of plastic items (Ye and Andrady, 1991, Lobelle and Cunliffe, 2011), none of them report the magnitude or speed of this process across different types of small fragments.
The plastics reported here were mostly white/transparent (84.7 %) or blue (8.3 %), which is consistent with reports from other investigations on buoyant marine plastics (Carpenter and Smith, 1972, Morét-Ferguson et al., 2010). Depending on the feeding ecology of the affected animal, ingested plastic colour proportions can differ from what is available in the environment (Schuyler et al., 2012). For instance, ingested plastic colour proportions in Australian shearwaters (*Ardenna pacifica* and *Ardenna tenuirostris*) are different from those reported by this study (Carey, 2011, Verlis et al., 2013). As these birds are known to use colour vision to select their food (Bowmaker, 1980, Verlis et al., 2013), colour can play a role in the ingestion risk associated with a certain plastic item. In contrast, the colour proportion of plastics found in scats of fur seals (*Arctocephalus* spp.) at Macquarie Island (Australia) reflected what was available as flotsam in this environment (Eriksson and Burton, 2003). These plastics are likely to be coming from the stomach contents of their main prey: the myctophid *Electrona subaspera*, which are pelagic small fish known to feed at night, selecting their food based on size rather than colour (Eriksson and Burton, 2003).

The vast majority (98.5 %) of the plastics detected were made of polyolefins (polyethylene and polypropylene), which is in agreement with what has been found for this size range of plastics in other marine regions around the world (Morét-Ferguson et al., 2010, Hidalgo-Ruz et al., 2012). Polyethylene and polypropylene account for most of our global plastic production (38 % and 24 %, respectively) (Andrady, 2011) and they are typically applied in the manufacturing of single-use disposable packaging. In addition to packaging, which reaches the oceans primarily from coastal areas, fishing equipment made of these polyolefins, e.g. fish crates, nets, ropes, fishing lines (Jones, 1995) are also likely sources of the plastic particles registered here. Other types of polymers found in this study include two pieces of expanded polystyrene (Styrofoam), a
type of plastic also used in packaging and fishing gear, and one fragment of ethylene vinyl acetate, which has several applications such as the making of shoe soles and foam mats.

2.5.2 Concentrations and sources

Our overall mean sea surface plastic concentration \( (Cs) \) was 4256.4 pieces km\(^{-2}\), which is similar to mean values reported for other regions outside subtropical gyres, such the Caribbean Sea (mean \( Cs = 1414 \) pieces km\(^{-2}\)) and Gulf of Maine (mean \( Cs = 1534 \) pieces km\(^{-2}\)) (Law et al., 2010). Within subtropical gyres, \( Cs \) values tend to be higher but within the same order of magnitude: 20328 pieces km\(^{-2}\) in the North Atlantic Gyre (Law et al., 2010), and 26898 pieces km\(^{-2}\) in the South Pacific Gyre (Eriksen et al., 2013). The exception seems to be the subtropical waters of the North Pacific and Mediterranean, which present mean \( Cs \) values that are an order of magnitude higher than those reported here: 116000 pieces km\(^{-2}\) in the Mediterranean (Collignon et al., 2012), 174000 pieces km\(^{-2}\) in Northwest Pacific (Yamashita and Tanimura, 2007), and 334271 pieces km\(^{-2}\) in Northeast Pacific (Moore et al., 2001). The latter is also known as the “Great Pacific Garbage Patch” (Moore et al., 2001), which is the largest aggregator of floating marine particles (van Sebille et al., 2012).

Our findings show that the distribution of marine plastics is quite widespread (93 % of our net stations had at least one plastic piece), patchy (i.e. high variability within and between net stations’ \( Cs \)) and dynamic (\( Cs \) ranged from 10% to 91% of \( Ci \)). Therefore, better spatio-temporal data coverage is required in order to identify plastic pollution hotspots within Australian waters. However, our data already indicate some spatial patterns: we observed high plastic concentrations close to Sydney and Brisbane cities. This suggests that plastics along Australia’s east coast are mostly associated with
domestic inputs. Since high quantities of plastic were also found close to Viti Levu (Fiji), we hypothesize that part of the plastics coming from coastal areas remain in the vicinity of their sources for a long time, while fragmenting into smaller pieces. This suggestion of local retention of plastic debris is in agreement with findings of recent studies, e.g. (Thiel et al., 2013), and could be tested by developing high-resolution models able to simulate plastic transport in coastal environments.

While the relatively high concentrations of plastic found close to the East Australian coast (net stations 18-20, 22, 37) seem to originate from local sources of plastics, those found in southwest Tasmania/eastern South Australia (net stations 5, 6, 8), and the North West Shelf (net station 54) could be associated with international sources and/or maritime operations. The presence of internationally-based plastics is suggested by (1) a fragment with Indonesian words that was collected in North West Shelf (see Figure 2.2) and (2) beach surveys, which registered in South Australia plastic debris from South Africa and South America (Edyvane et al., 2004). High plastic concentrations in the southern tip of Tasmania (net station 5) might be caused by convergence effects of the encounter of the East Australian and Zeehan coastal currents (Edyvane et al., 2004), whereas those found off the east coast (e.g. net station 37) could be associated with meso-scale eddies of the East Australian current (Ridgway and Dunn, 2003).

Aside from this study and the one that developed the biophysical model we applied here (Kukulka et al., 2012), we are not aware of any investigation that quantitatively considers the effect of vertical mixing processes on plastic concentrations. This effect needs to be taken into account in future studies assessing at-sea plastic pollution to allow better comparisons between data collected under different sea states. An important step towards improved simulations of plastic distribution along the water column is to better quantify the buoyant rise velocity ($w_b$) of plastic particles from
different oceanic and coastal surface waters. This variable has a considerable impact on the output of the model applied here. Furthermore, other environmental variables that were not taken into account in our one-dimensional column model (e.g. Langmuir circulation, breaking waves, mixed layer depth) could be incorporated in this type of modeling.

2.5.3 Potential pathways

The model outputs and routes taken by real drifters showed that plastics we found could have moved via a wide range of routes. This is because our net stations are within regions that experience different hydrodynamics, e.g. North West Shelf, Great Australian Bright, Coral Sea, and Tasman Sea (Schiller et al., 2008). Plastics have the potential to reach the sampled sites by travelling with a range of currents, including: (1) Antarctic Circumpolar current (Talley et al., 2011a), which can carry plastics from a wide area to several of our net stations, particularly those along the coast of Tasmania, south coast of Australia, and Tasman Sea (net stations 1-15, 38 and 39); (2) South Equatorial current in the Pacific Ocean (Webb, 2000, Talley et al., 2011a), which can bring international plastics to the east coast of Australia (net stations 16-24, 40-45, 36, 37) and areas close to Fiji and New Caledonia (net stations 25-35); (3) East Australian current (Ridgway and Dunn, 2003, Talley et al., 2011a), which can carry plastics from domestic highly populated regions (e.g. Brisbane and Sydney) to the net stations along the coast of Tasmania (net station 5, 15, 38, 39), east coast of Australia (net stations 16-24, 36, 37) and the Tasman Sea (net stations 1-4); (4) Holloway, Leeuwin, South Australian, and Zeehan coastal current systems (Ridgway and Condie, 2004, Sandery and Kämpf, 2007, Condie and Andrewartha, 2008, Pattiaratchi and Woo, 2009), which can bring plastics from international areas connected to the Indonesian Throughflow and Indian Gyre, e.g. Southeast Asia/Indonesia (Lebreton et al., 2012), as well as from
domestic populated areas, to the net stations of the North West Shelf (net stations 48-57), off Perth (net station 14), and along the south coast of Australia, Bass Strait, Tasman Sea, and coast of Tasmania (net stations 1-13, 15-17, 37-39); and (5) West Australia current (Pattiaratchi and Woo, 2009), which could transport international marine plastics that accumulated in the Indian Gyre to the net stations in the North West Shelf (net stations 48-57) and off Perth (net station 14).

It is important to note that running models backwards and using drifter trajectories arriving at sampled locations can only provide an indication of the directions that the collected plastics could have taken. To precisely estimate plastic pathways is quite challenging, mostly because plastic source locations and quantities are still largely unknown. Moreover, there are still no methods to estimate the “age” (drifting time) of a certain plastic particle. For instance, only plastics with long drifting times (years) could have matched the long tracks of drifters. Another limitation of the real drifter approach is that the resulting pathway formed by all drifter tracks arriving at a certain region is not only dependent on the ocean current systems, but also on the locations where most of the drifters were released. For instance, sampled sites in the North West Shelf (net stations 48-57) had only a few drifters arriving at them. This is mostly due to the non-existence of drifters in the shallow waters of the Indonesian archipelago. The creation of a shallow-water drifter - e.g. (Ohlmann et al., 2005) - release program in this area could bring crucial information to help inform marine plastic pathways and sources.

2.5.4 Final remarks

This investigation shows that the abundant and widespread small marine plastics around Australia are likely coming from a variety of domestic and international, land- and ocean-based sources. Even though marine plastic pollution is a global environmental
issue, mostly caused by our massive production of plastic single-use disposable items, there are still no attempts to regulate plastic disposal on land at an international level (Rochman et al., 2013a). Additionally, dumping of plastics into the oceans remains a significant issue owing to difficulties with regulation and enforcement (Jones, 1995, Rakestraw, 2012). We suggest further at-sea studies on the characterization, spatial distribution, and pathways of marine plastics in coastal and oceanic regions around Australia, as well as on marine plastic toxin loads and interactions between small plastic particles and organisms at all trophic levels of the food web. This would improve our current knowledge on the effects of plastic on the marine ecosystem as a whole.
Chapter 3  Millimetre-sized Marine Plastics: A New Pelagic Habitat for Microorganisms and Invertebrates

3.1 Summary

Millimetre-sized plastics are abundant in most marine surface waters, and known to carry fouling organisms that potentially play key roles in the fate and ecological impacts of plastic pollution. In this study we used scanning electron microscopy to characterize biodiversity of organisms on the surface of 68 small floating plastics (length range = 1.7 - 24.3 mm, median = 3.2 mm) from Australia-wide coastal and oceanic, tropical to temperate sample collections. Diatoms were the most diverse group of plastic colonizers, represented by 14 genera. We also recorded ‘epiplastic’ coccolithophores (7 genera), bryozoans, barnacles (Lepas spp.), a dinoflagellate (Ceratium), an isopod (Asellota), a marine worm, marine insect eggs (Halobates sp.), as well as rounded, elongated, and spiral cells putatively identified as bacteria, cyanobacteria, and fungi. Furthermore, we observed a variety of plastic surface microtextures, including pits and grooves conforming to the shape of microorganisms, suggesting that biota may play an important role in plastic degradation. This study highlights how anthropogenic millimetre-sized polymers have created a new pelagic habitat for microorganisms and invertebrates. The ecological ramifications of this phenomenon for marine organism dispersal, ocean productivity, and biotransfer of plastic-associated pollutants, remain to be elucidated.
3.2 Introduction

Millimetre-sized plastics resulting from the disintegration of synthetic products (known as ‘microplastics’ if smaller than 5 mm) are abundant and widespread at the sea surface (Moore et al., 2001, Law et al., 2010, Morét-Ferguson et al., 2010, Hidalgo-Ruz et al., 2012, Eriksen et al., 2013, Law et al., 2014). These small marine plastics are a toxic hazard to food webs since they can contain harmful compounds from the manufacturing process (e.g. Bisphenol A), as well as contaminants adsorbed from the surrounding water (e.g. polychlorinated biphenyls) (Mato et al., 2001, Rios et al., 2007, Teuten et al., 2009, Rochman et al., 2013a). These substances can be carried across marine regions and transferred from plastics to a wide range of organisms, from zooplankton and small fish to whales (Basheer et al., 2004, Teuten et al., 2009, Boerger et al., 2010, Fossi et al., 2012, von Moos et al., 2012, Gassel et al., 2013, Choy and Drazen, 2013, Cole et al., 2013, Rochman et al., 2013c). Furthermore, they can physically damage suspension- and deposit-feeding fauna (e.g. internal abrasions and blockages after ingestion) (Wright et al., 2013), and alter pelagic and sediment-dwelling biota by modifying physical properties of their habitats (Carson et al., 2011). Finally, these small marine plastics can transport rafting species (Carpenter and Smith, 1972, Carpenter et al., 1972, Gregory, 1978, Gregory, 1983, Carson et al., 2013, Zettler et al., 2013), potentially changing their natural ranges to become non-native species and even invasive pests.

Apart from providing long-lasting buoyant substrata that allow many organisms to widely disperse (Winston, 1982, Jokiel, 1990, Barnes, 2002, Masó et al., 2003, Barnes and Fraser, 2003, Aliani and Molcard, 2003, Barnes, 2004, Thiel and Gutow, 2005, Gregory, 2009, Fortuño et al., 2010, Goldstein et al., 2014), marine plastics may also supply energy for microbiota capable of biodegrading polymers and/or associated compounds (Sudhakar et al., 2007, Artham and Doble, 2009, Balasubramanian et al.,
2010, Harrison et al., 2011, Zettler et al., 2013, Harshvardhan and Jha, 2013), and perhaps for invertebrates capable of grazing upon plastic inhabitants. The hydrophobic nature of plastic surfaces stimulates rapid formation of biofilm, which drives succession of other micro- and macro-organisms. This ‘epiplastic’ community appears to influence the fate of marine plastic pollution by affecting the degradation rate (Andrady, 2011, Zettler et al., 2013), buoyancy (Ye and Andrady, 1991, Moore et al., 2001, Lobelle and Cunliffe, 2011), and toxicity level (Harrison et al., 2011) of plastics. Moreover, epiplastic microbiota could have impacts on the microflora of its consumers, and infectious organisms may reach their hosts through plastic ingestion (Harrison et al., 2011, Pham et al., 2012, Zettler et al., 2013).

Although epiplastic organisms may play an important role in determining the fate and ecological impacts of plastic pollution, little research has been directed to such study, particularly on the inhabitants of the widely dispersed and abundant millimetre-sized marine plastics (Harrison et al., 2011). In 1972, two papers first reported the occurrence of organisms (diatoms, hydroids, and bacteria) on small plastics (0.1 – 5 mm long) collected by plankton nets (Carpenter and Smith, 1972, Carpenter et al., 1972). Further at-sea studies focusing on microplastic fouling biota only emerged in the 2000s (Carson et al., 2011, Goldstein et al., 2012, Zettler et al., 2013). Zettler et al. (2013) conducted the first comprehensive characterisation of epiplastic microbial communities, which they coined the “Plastisphere”. These authors used scanning electron microscopy (SEM) and next-generation sequencing to analyse three polyethylene and three polypropylene plastic pieces (approx. 2 – 20 mm long) from offshore waters of the North Atlantic. This pioneer study revealed a unique, diverse, and complex microbial community that included diatoms, ciliates, and bacteria.
Here, we used SEM to examine types of organisms inhabiting the surface of 68 small marine plastics (length range = 1.7 - 24.3 mm, median = 3.2 mm) from inshore and offshore waters from around the Australian continent (Figure 3.1). We contributed many new records of taxa associated with millimetre-sized marine plastics and imaged a variety of marine plastic shapes and surface textures resulting from the interaction of polymers with environments and organisms.

![Figure 3.1 Sampling locations of the 68 plastics analysed in this study.](image)

Black lines delimit marine regions of Australia (environment.gov.au/topics/marine/marine-bioregional-plans); dots indicate areas where the analysed plastics were collected; numbers represent how many plastics were taken for scanning electron microscopy analyses at these locations. Samples collected were fragments of hard plastic (N = 65), except at locations marked with an asterisk: one piece of Styrofoam cup in Fijian waters, one pellet in South Australia, and one piece of soft plastic in the Australia’s North-west marine region.

### 3.3 Material and Methods

Ethics Statement: Permits to conduct field research within the Great Barrier Reef area were obtained from the Great Barrier Reef Marine Park Authority (GBRMPA: permit
G11/34378.1). No other special permit was required since sampling was limited to marine debris.

Buoyant plastics were collected using surface net tows in waters around Australia (see details in Chapter 2) and preserved in 2.5% glutaraldehyde buffered in filtered seawater. Prior to analysis with a scanning electron microscope, plastics were dehydrated through a series of increasing ethanol concentrations (up to 100%), critical-point dried using CO₂, mounted on aluminium stubs with carbon tape, and sputter coated with a 20 - 30 nm layer of gold. We used a Zeiss 1555 VP-FESEM scanning electron microscope operated at 10 - 20 kV, 11 - 39 mm working distance, and 10 - 30 μm aperture. We randomly selected 65 hard plastics among those small enough to fit onto SEM stubs (< 10 mm) and large enough to be easily handled (> 1 mm). For comparison, a piece of (1) soft plastic, (2) industrial plastic pellet, and (3) expanded polystyrene (Styrofoam) were also examined, totalling 68 plastic pieces examined with SEM. These plastics were collected from offshore waters of the South-west Pacific (N = 19) and from different Australian marine regions (environment.gov.au/topics/marine/marine-bioregional-plans): North-west (N = 13), South-west (N = 3), South-east (N = 13), Temperate East (N = 16), and Coral Sea (N = 4; Figure 3.1).

The different types of organisms detected on each plastic piece were imaged, measured using ImageJ (length and width, http://rsb.info.nih.gov/ij/), classified into taxonomic/morphological groups, and the frequency of occurrence (FO) for each type was calculated. We used online resources, e.g. marinespecies.org, westerndiatoms.colorado.edu, primary taxonomic literature, e.g (Cheng, 1976, Weise and Rheinheimer, 1977, Kensley, 1994, Hallegraeff et al., 2010, Magalhães and Bailey-Brock, 2012), and expert consultation to identify the organisms at the lowest possible
taxonomic level. Long filaments were very common but were excluded from the analysis due to difficulty in determining if they were organisms or mucilage.

For each plastic piece observed, an image of the entire piece was taken at 50x magnification. These images were uploaded to ImageJ to measure plastic particles’ size parameters - length, area, perimeter, aspect ratio, and shape parameters - circularity and solidity indexes (Paulrud et al., 2002, Ferreira and Rasband, 2012). Surface fractures, pits and grooves (Corcoran et al., 2009, Cooper and Corcoran, 2010) were also observed, recorded, and imaged while examining the entire surface of the plastics at magnifications of 100 – 500x. Other peculiar microtextures observed at higher magnifications, such as those suggesting interactions with biota, were also recorded and imaged. After SEM analyses, plastics were washed with distilled water and submitted to Fourier Transform Infrared spectrometry (FT-IR) for polymer identification. Two plastic pieces were destroyed while being cleaned for FT-IR; as such, we identified the polymer of 66 out of the 68 plastics examined using SEM.

3.4 Results

We examined 65 hard plastic fragments with lengths ranging from 1.7 to 8.9 mm (median = 3.2 mm), one 4 mm-wide plastic pellet, one 8.7 mm portion of a 15 mm long soft plastic fragment, and one 7 mm piece of a 24.3 mm Styrofoam cup fragment. Apart from the Styrofoam cup fragment (expanded polystyrene), plastics were made of polyethylene (N = 54) and polypropylene (N = 11). Hard plastics had a diverse range of shapes (solidity index = 0.87 – 0.98, circularity index = 0.28 – 0.83; Figure 3.2) and types of surface microtextures, including linear fractures, pits, and scraping marks (Appendix 3). Diatoms and bacteria (rounded, and elongated cells) were by far the most frequently observed organisms, being detected in all sampled marine regions (Figure
3.3). Plastics’ FT-IR spectra, 1143 SEM micrographs, and a matrix containing information from collection sites, plastics characteristics, and organism/microtexture presence-absence data are available in (Reisser et al., 2014).
Figure 3.2 Overall appearance of marine plastics, as shown by scanning electron micrographs. Dot colour indicates the marine region where the piece was sampled (see legend and Figure 3.1). Pieces are hard plastic fragments, with the exception of the soft plastic fragment (red dot), pellet (yellow dot), and Styrofoam fragment (green dot) shown at the bottom of the diagram and marked with a white asterisk. All images are at the same magnification (see scale bar at lower right).
Figure 3.3 Types of epiplastic organisms detected at each of the marine regions sampled in this study (see Figure 3.1). Lines connect types of organisms (squares) to the marine regions (circles) where they were observed. Line colour indicates type of organism, with black lines representing invertebrates. Line thickness is proportional to the organism’s frequency of occurrence (FO = <25%, 25-50%, 50-75%, >75%).

Diatoms were the most abundant, widespread, and diverse group of plastic colonizers (Figure 3.3 and Figure 3.4). These organisms were frequently observed (FO = 78%, N = 68 plastics) and included symmetrical biraphids/naviculoids (**Navicula** subgroup...
lineatae, *Mastogloia* sp., *Haslea* sp.; Figure 3.4a-c), Nitzschioids (*Nitzschia* spp., *Nitzschia longissima*; Figure 3.4d-f), monoraphids (*Cocconeis* spp., *Achnanthes* sp.; Figure 3.4g-i), centrics (*Minidiscus trioculatus*, *Thalassiosira* sp.; Figure 3.4j), araphids (*Thalassionema nitzschioides* var. *parva*, *Microtabella* spp., *Licmophora* spp., *Grammatophora* sp.; Figure 3.4k,l,o), and asymmetrical biraphids (*Amphora* spp., *Cymbella* sp.; Figure 3.4m,n). Most diatoms were growing flat on the surface (adnate and motile diatoms), but some were erect, attached to plastics by mucous pads or stalks/peduncles. The genus *Nitzschia* was the most frequent diatom (FO = 42.6%), followed by *Amphora* (13.2%), *Licmophora* (11.8%), *Navicula* (8.8%), *Microtabella* (5.9%), *Cocconeis* (4.4%), *Thalassionema* (2.9%), and *Minidiscus* (2.9%). The other six genera were only detected on a single plastic piece (FO = 1.5%). These frequencies of occurrence are likely to be underestimated, as many diatoms could not be identified from girdle-view positions (FO unidentified diatoms = 45.6%).
Figure 3.4 Examples of epiplastic diatoms.

a: *Navicula* sp.; b: *Mastogloia* sp.; c: small naviculoids; d: *Nitzschia* sp.; e: *Nitzschia* sp.; f: *Nitzschia longissima*; g,h: *Cocconeis* spp.; i: *Achnanthes* sp.; j: *Thalassiosira* sp.; k: *Thalassionema nitzschioides*; l: *Microtabella* sp.; m,n: *Amphora* spp.; o: *Licmophora* sp.

Calcereous coccolithophores were observed only on plastics from southern Australia (South-east and South-west marine regions; FO = 37.5%, N = 16 plastics; Figure 3.3,
Figure 3.5a-h). The species identified included *Calcidiscus leptoporus* (Figure 3.5a), *Emiliania huxleyi* (Figure 3.5b,c), *Gephyrocapsa oceanica* (Figure 3.5d), *Umbellosphaera tenuis* (Figure 3.5e), *Umbilicosphaera hulburtiana* (Figure 3.5f), *Coccolithus pelagicus* (Figure 3.5g), and *Calciosolenia* sp. (Figure 3.5h). Many of these observations related to detached coccolith scales held in place by mucilage and chitin filaments resembling those produced by diatoms (e.g. *Thalassiosira*; Figure 3.5b,f). However, intact coccospheres were also present (Figure 3.5c,d,f). Additionally, one specimen of the dinoflagellate *Ceratium* cf. *macroceros* was present on a 8.2mm plastic from South-west Australia (Figure 3.3, Figure 3.5i).
Figure 3.5 Examples of epiplastic coccoliths (a-h) and dinoflagellate (i).

a: Calcidiscus leptoporus; b, c: Emiliania huxleyi; d: Gephyrocapsa oceanica; e: Umbellosphaera tenuis; f: Umbilicosphaera hulburtiana; g: Coccolithus pelagicus; h: Calciosolenia sp.; i: Ceratium cf. macroceros.
We found several unidentified organisms of various morphotypes and sizes, mostly resembling bacterial, cyanobacterial, and fungal cells (Figure 3.6). After diatoms, rounded/oval cells (length-width ratio < 1.5; Figure 3.6a-c,i-m) were the most frequently observed morphotype (FO = 72%, N = 68 plastics; Figure 3.3). Rounded/oval cells with widths < 1 μm and ≥ 1 μm had an overall FO of 38.2% and 54.4%, respectively.

Elongated cells (length-width ratio ≥ 1.5; Figure 3.6e-h) were also frequently observed, being detected on 59% of the plastics examined (Figure 3.3). Those with widths < 1 μm and ≥ 1 μm had an overall FO of 51.5% and 11.7%, respectively. Spiral cells (Figure 3.6d) had similar appearances (resembling spirochaete bacteria) and sizes (0.2 - 0.3 μm width), and were only observed in the South-west Pacific region (FO = 31.6%, N = 19; Figure 3.3). Several plastic pits and grooves contained bacteria-like cells closely resembling their shape (Figure 3.6i-m). They were particularly common on plastics covered by large rounded cells (Figure 3.6k).
Figure 3.6 Examples of epiplastic rounded, elongated and spiral cells.
a, b, c: rounded cells; d: spiral “spirochaete” cell; e, f, g, h: elongated cells.; i, j, k, l, m: pits and grooves on plastics with rounded cells.

A few invertebrates were observed on the millimetre-sized plastics (FO = 16.2%, N = 68 plastics; Figure 3.3 and Figure 3.7). Colonies of encrusting bryozoans were the most common epiplastic animal (FO = 8.8%; Figure 3.7a-d). They occurred on two fragments from the Temperate East marine region and on four fragments from oceanic waters of the South-west Pacific (plastic length = 3.2 - 5.4 mm). Four of these bryozoan colonies were hosting abundant diatom assemblages dominated by *Licumophora* sp., *Nitzschia longissima* (Figure 3.7a), *Amphora* sp. (Figure 3.7c), and *Nitzschia* sp. (Figure 3.7d).
Additionally, lepadomorph barnacles (*Lepas* spp.; Figure 3.7e,f) were attached to the 24.3 mm Styrofoam cup fragment and to a 8.2 mm-long hard plastic; an Asellote isopod (Figure 3.7g) was found on the Styrofoam cup fragment; eggs of the marine insect *Halobates* sp. (Figure 3.7h) were observed on two plastics (4.6 and 5.5 mm long); and a unidentified marine worm (Figure 3.7i,j) was found on a 6 mm hard plastic fragment.
Figure 3.7 Examples of epiplastic invertebrates.

a: Bryozoan colony harboring an abundant assemblage of *Nitzschia longissima* (zoomed image shows part of this assemblage, scale bar = 20 μm); b: bryozoan colony relatively free of fouling; c: bryozoan-plastic interface displaying an abundant epizoic assemblage of *Amphora* sp.; d: bryozoan-plastic interface displaying an abundant epizoic assemblage of *Nitzschia* sp.; e, f: barnacles (*Lepas* spp.); g: Asellota isopod; h: egg of the marine insect *Halobates* sp.; i: marine worm; j: zoom on the surface of the unidentified marine worm shown in ‘i’. 
3.5 Discussion

There now exists a large body of evidence that millimetre-sized plastics are abundant and widespread in marine environments (Carpenter et al., 1972, Carpenter and Smith, 1972, Moore et al., 2001, Law et al., 2010, Morét-Ferguson et al., 2010, Hidalgo-Ruz et al., 2012, Eriksen et al., 2013) and our study significantly adds to this by conclusively demonstrating that they are colonized by a wide range of biota, particularly diatom and bacteria species - Table 1 (Carpenter and Smith, 1972, Gregory, 1978, Gregory, 1983, Moore et al., 2001, Carson et al., 2013, Zettler et al., 2013). We more than doubled the number of known diatom genera inhabiting millimetre-sized marine plastics and provide the first identifications of coccolithophore genera attached to these floating plastic particles. We also recorded a few invertebrate species living on these small plastics. As such, our findings provide further evidence that not only large debris (Winston, 1982, Jokiel, 1990, Aliani and Molcard, 2003, Barnes, 2002, Barnes and Fraser, 2003, Masó et al., 2003, Barnes, 2004, Thiel and Gutow, 2005, Gregory, 2009, Fortuño et al., 2010, Goldstein et al., 2014) serve as vehicles for organism dispersal. Abundant ‘microplastics’ are equally providing a new pelagic habitat to many microorganism and a few invertebrate taxa.
Table 1 List of known genera occurring on millimetre-sized pelagic plastics.

Organism groups (first column), their abundance and/or frequency of occurrence (when available; second column), and genera (third column). References are indicated by superscript letters and given at the bottom of the table, along with approximate length range of plastics examined. Genera in bold indicate those first detected in this study.

<table>
<thead>
<tr>
<th>Group</th>
<th>Abundance/FO</th>
<th>Genera</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bacteria</td>
<td>1833 per mm^2</td>
<td>Acinetobacter^b, Albidovulum^b, Alteromonas^b, Amoebophilus^b, Bacteriovorax^b, Bdellovibrio^b, Blastopirellula^b, Devisia^b, Erythrobacter^b, Filomicrobium^b, Fulvivirga^b, Haliscenomobacter^b, Hellea^a, Henriciella^b, Hyphomonas^b, Idiomarina^b, Labrenzia^b, Lewinella^b, Marinocellum^b, Microscilla^a, Muricauda^a, Nitrotireductor^b, Oceaniserpentilla^b, Parvularcula^a, Pelagibacter^b, Phycisphaera^b, Phormidium^b, Pleurocapsa^b, Prochlorococcus^b, Pseudoalteromonas^b, Pseudomonas^b, Psychrobacter^b, Rhodovulum^b, Rivularia^b, Roseovarius^b, Rubrimonas^b, Sediminibacterium^b, Synechococcus^b, Thalassobius^b, Thiobios^b, Tenacibaculum^b, Thalassobius^b, Vibrio^b</td>
</tr>
<tr>
<td>Diatoms</td>
<td>77.9%</td>
<td>Amphora^a, Achananthes^a, Chaetoceros^a, Cocconeis^a, Cyclotella^c, Cymbella^a, Grammatophora^a, Haslea^a, Licmophora^a, Mastogloia^a, Microtabella^a, Minidiscus^a, Navicula^b, Nitzschia^a, Pleurosigma^a, Sellaphora^a, Stauroneis^b, Thalassionema^a, Thalassiosira^a</td>
</tr>
<tr>
<td>Coccoliths</td>
<td>8.8%</td>
<td>Calcidiscus^a, Emiliana^a, Gephyrocapsa^a, Umbellosphaera^a, Umbilicosphaera^a, Coccolithus^a, Calciosolenia^a</td>
</tr>
<tr>
<td>Bryozoa</td>
<td>8.8%</td>
<td>Membranipora^f, Jellyella^f, Bowerbankia^f, Filicrisia^a</td>
</tr>
<tr>
<td>Hydroids</td>
<td></td>
<td>Clytia^c, Gonothyracea^c, Obelia^c</td>
</tr>
<tr>
<td>Polychaete</td>
<td></td>
<td>Spirorbis^a, Hydroidea^a</td>
</tr>
<tr>
<td>Dinoflagellates</td>
<td>1.5%</td>
<td>Alexandrium^b, Ceratium^a</td>
</tr>
<tr>
<td>Insect eggs</td>
<td>2.9%</td>
<td>Halobates^a,b</td>
</tr>
<tr>
<td>Barnacles</td>
<td>2.9%</td>
<td>Lepas^a</td>
</tr>
<tr>
<td>Rhodophyta</td>
<td></td>
<td>Flosiella^b</td>
</tr>
<tr>
<td>Foraminifer^a</td>
<td></td>
<td>Discorbis^g</td>
</tr>
<tr>
<td>Radiolaria</td>
<td></td>
<td>Circorhchema^d</td>
</tr>
<tr>
<td>Ciliate</td>
<td></td>
<td>Ephelota^b</td>
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</tbody>
</table>

This study (1.7 – 24.3 mm), Zettler et al. 2013 (2 – 20 mm), Carpenter and Smith 1972 (2.5 – 5 mm), Carson et al. 2013 (1 – 10 mm), Goldstein et al. 2014 (4 – 5 mm), Gregory 1978 (2 – 5 mm), Gregory 1983 (1 – 5 mm), Majer et al. 2012 (2 – 5 mm), Goldstein et al. 2012 (1.2 – 6.5 mm), Carpenter et al. 1972 (0.1 – 2 mm)

We observed fouling diatoms to be diverse and widespread on marine plastics. These diatoms seemed to firmly attach to the plastic, resisting water turbulence and wave action. All the identified diatom genera are well known to form biofilms on estuarine and marine sediments and rocks (epilithic), vegetation (epiphytic), and animals...
(epizoic) (Carpenter, 1970, Totti et al., 2009, Reisser et al., 2010, Congestri and Albertano, 2011, Tiffany, 2011, Romagnoli et al., 2014); marine plastics thus create a novel, long-lasting and abundant floating habitat for ‘benthic’ diatoms, in a light and nutrient-filled environment that is stable and beneficial to these organisms. Future epiplastic diatom research should focus on the quantitative contribution of these organisms to enhancing primary and secondary productivity of different marine regions, such as within subtropical gyres where productivity tends to be low but plastic pollution level high (Moore et al., 2001, Polovina et al., 2008, Law et al., 2010, Eriksen et al., 2013). Because of their rapid growth and production of extracellular substances (Kawamura et al., 1995), epiplastic diatoms may provide an important food source for invertebrate grazers. As plastic debris can contain harmful substances (Mato et al., 2001, Rios et al., 2007, Teuten et al., 2009, Gassel et al., 2013, Rochman et al., 2013c), it remains unclear if such grazer-plastic relationships would have a positive or negative impact on the populations involved in this new type of food web.

A significant number of coccolithophore species were present on millimetre-sized marine plastics. These planktonic organisms are not commonly recognized as fouling or rafting organisms (Thiel and Gutow, 2005), although their occasional occurrence on marine plastics was briefly mentioned in recent studies (Carson et al., 2013, Zettler et al., 2013). Some of our observations were of clusters of mixed coccolith species, resembling zooplankton fecal pellets, and of solitary coccoliths, likely detached from living coccocosomes and stuck to clingy parts of the plastic biofilm. However, entire coccolithophores were also seen attached to plastics, suggesting that these organisms could be using ocean plastics as ‘floating devices’. We only observed coccoliths on plastics from southern Australia; as such, additional studies in these temperate waters may help better understand this potential coccolith-plastic relationship. Another atypical
organism detected was the planktonic dinoflagellate Ceratium cf. macroceros. Recent studies have found plastics heavily fouled by dinoflagellates, including individuals and cysts of the potentially harmful species Ostreopsis sp., Coolia sp., and Alexandrium spp. (Masó et al., 2003, Zettler et al., 2013), but here we only detected a single specimen of this group.

Several unidentified organisms (rounded, oval, elongated, and spiral) resembling bacterial cells were flourishing on millimetre-sized marine plastics. This supports previous studies that describe well established bacterial populations growing on plastic fragments (Carson et al., 2013, Zettler et al., 2013). Many of these unidentified cells were apparently interacting with the plastic surface by forming pits and grooves. Within this group of “pit-formers”, colonies of rounded cells (around 5 micron in diameter) covered large areas of the plastic surface. They were similar to some previously unidentified epiplastic organisms from the North Atlantic (Zettler et al., 2013). These SEM observations, along with detections of putative hydrocarbon-degrading bacteria on marine plastics (Zettler et al., 2013) and experiments demonstrating that marine bacteria can biodegrade polymers (Sudhakar et al., 2007, Artham and Doble, 2009, Balasubramanian et al., 2010, Harrison et al., 2011, Zettler et al., 2013, Harshvardhan and Jha, 2013), strongly suggest that plastic biodegradation is occurring at the sea surface. Such process could partially explain why quantities of millimetre-sized marine plastics are not increasing as much as expected (Law et al., 2010, Law et al., 2014). Studies of the “Plastisphere” from different marine regions worldwide will prove invaluable for extending our knowledge on epiplastic marine microbial communities, and may support the development of biotechnological solutions for better plastic waste disposal practices (Sivan, 2011, Sangale et al., 2012, Webb et al., 2012).
A number of invertebrates inhabited the small plastics examined here: bryozoans, barnacles *Lepas* spp., an Asellota isopod, a marine worm, and eggs of the marine insect *Halobates* sp. Even though microplastic-associated animals are rare and less diverse when compared to those associated with macroplastics (Winston, 1982, Jokiel, 1990, Barnes, 2002, Aliani and Molcard, 2003, Barnes and Fraser, 2003, Masó et al., 2003, Barnes, 2004, Thiel and Gutow, 2005, Gregory, 2009, Fortuño et al., 2010, Goldstein et al., 2014), ecological implications of this phenomenon may be significant, e.g. (Goldstein et al., 2012), given the large quantities and wide distribution ranges of millimetre-sized plastics in the marine environment (Carpenter et al., 1972, Carpenter and Smith, 1972, Moore et al., 2001, Law et al., 2010, Morét-Ferguson et al., 2010, Hidalgo-Ruz et al., 2012, Eriksen et al., 2013). Among the effects plastic associates may have is to shape ‘epiplastic’ microbiota by hosting unique epizoic assemblages on their bodies. For instance, the bryozoan colonies examined here covered a large proportion of their plastic-host, with some of them harboring unique diatom-dominated assemblages. Previous studies have shown that bryozoans do not represent neutral surfaces for microbial colonizers (Scholz and Hillmer, 1995, Kittelmann and Harder, 2005), with some species offering a favourable habitat for diatoms when compared to the surrounding substratum, e.g. by protecting against predators and supplying nutrients through flow generated by polypids (Wuchter et al., 2003). Further studies focusing on both epiplastic microorganisms and invertebrates have the potential to further elucidate symbiotic and/or competitive relationships between inhabitants of this new type of pelagic habitat.

In summary, this study showed that millimetre-sized marine plastics are providing a new niche for several types of microorganisms and some invertebrates. This phenomenon has considerable ecological ramifications and deserves further research.
As discussed here, additional observational and experimental studies on the inhabitants of these small plastic fragments may better elucidate several key plastic pollution processes that remain poorly assessed, such as at-sea polymer degradation and mineralisation, impacts of epiplastic communities on their consumers, and changes in the distributional range of species by plastic rafting.
Chapter 4  The vertical distribution of buoyant plastics at sea: an observational study in the North Atlantic Gyre

4.1 Summary

Millimeter-sized plastics are numerically abundant and widespread across the world’s ocean surface. These buoyant macroscopic particles can be mixed within the upper water column due to turbulent transport. Models indicate that the largest decrease in their concentration occurs within the first few meters of water, where *in situ* observations are very scarce. In order to investigate the depth profile and physical properties of buoyant plastic debris, we used a new type of multi-level trawl at 12 sites within the North Atlantic subtropical gyre to sample from the air-seawater interface to a depth of 5 m, at 0.5 m intervals. Our results show that plastic concentrations drop exponentially with water depth, and decay rates decrease with increasing Beaufort scale. Furthermore, smaller pieces presented lower rise velocities and were more susceptible to vertical transport. This resulted in higher depth decays of plastic mass concentration (milligrams m$^{-3}$) than numerical concentration (pieces m$^{-3}$). Further multi-level sampling of plastics will improve our ability to predict at-sea plastic load, size distribution, drifting pattern, and impact on marine species and habitats.
4.2 Introduction

Plastics pose physical and chemical threats to the oceans’ ecosystem. Their widespread occurrence at the sea surface may be shifting the distribution and abundance of marine populations due to (1) enhanced ocean drift opportunities and (2) damaging effects on biota and habitats. Plastics harbour organisms - such as fouling microorganisms, invertebrates, and fish - that can widely disperse via this new type of habitat, potentially entering non-native waters (chapter 3) (Winston et al., 1997, Barnes, 2002, Thiel and Gutow, 2005, Zettler et al., 2013). Plastic objects can also entangle or be ingested/inhaled by marine animals, leading to impacts such as starvation, death, and hepatic stress (Derraik, 2002, Browne et al., 2008, Gregory, 2009, Rochman et al., 2013c, Watts et al., 2014).

Most of what is known about at-sea plastic characteristics and concentrations comes from surface net sampling, where the top few centimetres of the water column is filtered to collect plastics larger than 0.2–0.4 mm (Hidalgo-Ruz et al., 2012). These sea surface samples have shown that the world’s sea surface contains many millimetre-sized plastic pieces known as ‘microplastics’ when smaller than 5 mm in length (Arthur et al., 2009, Hidalgo-Ruz et al., 2012). This type of plastic pollution is widespread across oceans, with higher contamination levels at convergence zones such as those within subtropical gyres (Carpenter and Smith, 1972, Maximenko et al., 2012, Lebreton et al., 2012, van Sebille et al., 2012, Cózar et al., 2014, Eriksen et al., 2014). Plastic debris collected by surface nets are mostly fragments of packaging and fishing gear made of polyethylene and polypropylene (chapter 2) (Barnes et al., 2009, Morét-Ferguson et al., 2010, Hidalgo-Ruz et al., 2012). These two resins are less dense than seawater and account for approximately 62% of the plastic volume produced each year (Andrady, 2011).
Turbulence in the upper-ocean layer can vertically mix buoyant plastic particles. A model developed by Kukulka et al. 2012 predicted that the largest decrease in plastic concentration occurs over the first meters of the water column, where only a few low-resolution measurements exist (Lattin et al., 2004, Doyle et al., 2011, Kukulka et al., 2012, Isobe et al., 2014). As studying ocean turbulent transport is heavily dependent on observations (Ballent et al., 2012, D'Asaro, 2014), high-resolution multi-level plastic sampling is needed to test this prediction. A better understanding of the vertical transport of buoyant plastics is fundamental for improving estimates of concentration, size distribution, and dispersal of plastics in the world’s ocean (chapter 2) (Kukulka et al., 2012, Law et al., 2014, Isobe et al., 2014).

In this context, the present study aimed at obtaining depth profiles of plastic pollution in the top layer of the oceans (0-5 m). We performed multi-level sampling with a new type of equipment to (1) quantify the exponential decay rates of plastic mass and numerical concentration with depth, and (2) demonstrate how these vary with sea state. We also provide the first experimental measurements of the rise velocity of plastic pieces, evaluating its relation to the type and size of pieces.

4.3 Materials and Methods

4.3.1 At-sea sampling

We conducted 12 multi-level net tows that sampled the upper 5 meters of the North Atlantic accumulation zone (Law et al., 2010, Maximenko et al., 2012, Lebreton et al., 2012) during day hours, from 19 to 22 May 2014, aboard the sailing vessel Sea Dragon (Figure 1). We used a new collection device capable of sampling surface waters from the air-seawater interface to a depth of 5 m, at 0.5 m intervals. This equipment is composed of eleven frames with 0.5 m height x 0.3 m width fitted with 2.1 m-long 150
µm mesh polyester nets. These nets were stacked vertically and secured within an external frame that was dragged in the water from eight towing points, ensuring its stability and perpendicular position in relation to the sea surface, with the top net completely above mean water line (see Figure 1). Tow durations ranged from 55 to 60 minutes and were all undertaken while the vessel was travelling at a speed of 1–1.9 knots. The captain, who has 20 years sailing experience, estimated wind speeds and sea state of each sampling period: Beaufort scale 1 (N = 3 net tows), 3 (N = 4 net tows), and 4 (N = 5 net tows) (Reisser et al., 2015). After each tow, we transferred the collected contents to a 150 µm sieve and stored them in aluminium bags that were kept frozen during transportation.

Figure 4.1 North Atlantic map indicating locations sampled during this study (orange dots) using the multi-level net displayed in the right panel.

The map also shows the expedition departure and arrival location (Bermuda), plastic accumulation zones as predicted by ocean modelling (Lebreton et al., 2012, Maximenko et al., 2012), and a surface net tow dataset (grey dots) (Law et al., 2010).
4.3.2 Estimating depth profiles of plastic contamination

We calculated plastic numerical and mass concentrations by dividing the number of plastic pieces and total plastic mass by the volume of filtered seawater of each net sample (pieces m\(^{-3}\) and milligrams m\(^{-3}\)). Filtered volume was estimated using frame dimensions and readings from a mechanical flowmeter (32 cm per rotation).

Samples were washed into a clear plastic container filled with filtered seawater, and floating macroscopic plastics were organised into gridded petri dishes for counting and characterisation. The searches for plastic pieces were of at least one hour per sample, with the aid of thumb forceps, dissecting needles, magnifying glasses, and LED torches. The latter was particularly important for detecting thin transparent plastic fragments, which had low detection probability when not reflecting light. Two thin filaments resembling textile fibres were discarded due to potential air contamination as noted in (Foekema et al., 2013). Once all plastics were counted and characterised, they were washed with deionised water, transferred to aluminium dishes, dried at 60° C, and weighed.

To quantify the variation of plastic concentration with depth and assess the effect of changing sea state on these vertical profiles, we first divided plastic concentration of samples by their corresponding surface concentration value. We then took the average of these normalised concentrations between adjacent nets to estimate normalised plastic concentration values at depths of: 0 m (top 2 nets), 1 m (3\(^{rd}\) and 4\(^{th}\) nets), 2 m (5\(^{th}\) and 6\(^{th}\) nets), 3 m (7\(^{th}\) and 8\(^{th}\) nets), 4 m (9\(^{th}\) and 10\(^{th}\)), and 4.75 m (11\(^{th}\) net). Finally, numerical and mass concentration values from tows collected under the same Beaufort scale were grouped and fitted to exponential decay models of the form \(N = e^{-\lambda z}\), where \(N = \) normalised plastic concentration, \(z = \) depth, and \(\lambda = \) decay rate.
We also predicted normalised plastic concentration depth profiles using the model described in Kukulka et al. (2012): $N = e^{\text{ezwbA} \cdot 0 - 1}$, where $z =$ depth, $w_b =$ plastic rise velocity, and $A_0 = 1.5 u_w k H_s$ with $u_w =$ frictional velocity of water, $k =$ 0.4 (von Karman constant), and $H_s =$ significant wave height. We considered $w_b = 0.0053$ m s$^{-1}$ (plastics’ median rise velocity, as estimated in this study), $H_s =$ 0.1 m, 0.6 m, or 1m (typical wave heights experienced at Beaufort scales 1, 3, and 4, respectively), and used the wind ranges of Beaufort 1, 3, and 4 (1-3 knots, 7-10 knots, and 11-16 knots, respectively) to estimate their respective $u_w$ values through the approximation proposed by (Pugh, 1987): $u_w = 0.00012 W_{10}$, where $W_{10} =$ ten-metre wind speed in m/s. Thus, the considered numerical ranges of frictional velocity of water ($u_w$) were: 0.0006-0.0019 m s$^{-1}$ for Beaufort scale 1, 0.0043 – 0.0062 m s$^{-1}$ for Beaufort scale 3, and 0.0068-0.0099 m s$^{-1}$ for Beaufort scale 4.

### 4.3.3 Characterising plastic length, type, resin, and rise velocity

We measured the length of all plastic pieces using a transparent ruler (0.5 mm resolution), and classified them into the following types: hard plastic - fragments of rigid plastic; sheet - fragments of thin plastic, with some degree of flexibility; line - fragments of fishing lines or nets; foam - expanded polystyrene fragments; and pellet - raw material used to produce plastic items (Fotopoulou and Karapanagioti, 2012). We also identified the resin composition of 60 pieces using Raman spectroscopy (WITec alpha 300RA+), and measured the rise velocity of 0-3 plastics from each sample collected.

Our method of rise velocity measurement is an adaptation of an experiment to examine the fall velocity of various types of sediment particles in different fluids (Allen, 1985). Firstly, we made two marks 12.5 cm from the ends of a 1 m long clear plastic tube
(diameter = 40 mm). Secondly, we filled the tube with filtered seawater, capped both its ends with rubber stops, and locked it in place with a clamp. One of the tube ends was then opened, a plastic piece placed inside, the tube closed again (with no trapped air), and quickly turned upside down and locked in place with a clamp, using a spirit level to adjust its vertical position. Finally, we recorded the time taken for the plastic piece to rise from one mark to the other (distance = 75 cm) using a stopwatch. This was measured 4 times per plastic piece, and the average was used as the estimation of its rise velocity ($w_b$). Rise velocities of different plastic types were separately plotted against plastic lengths ($l$), and linear regressions of the form $w_b = al + b$ were applied to assess the effect of plastics’ characteristics on its rise velocity. We also plotted the rise velocities of plastic pieces collected at different depths to visualise depth patterns.

Finally, we calculated the fractions of plastics of different size classes (0.5-1 mm, 1.5-2 mm, 2.5-3 mm, 3.5-4 mm, 4.5-5 mm, > 5.5 mm) that were located at the sea surface (depth < 0.5 m) and in deeper layers (depth > 0.5 m) during sampling at Beaufort scales 1, 3, and 4. We calculated these fractions using all plastics collected, as well as separated by plastic type.

4.4 Results

4.4.1 Profiles of mass and numerical concentrations

Plastic numerical and mass concentrations both decreased abruptly from their peak values at the sea surface, where median values were equal to 1.69 pieces m$^{-3}$ and 1.60 mg m$^{-3}$ (Figure 4.2). Concentration differences between surface and deeper layers were higher in terms of mass than number of particles. For instance, median mass and numerical concentrations at 0.5-1 m were respectively 13.3 and 6.5 times lower than their median plastic peaks at 0-0.5 m.
Exponential models fitted well with both numerical and mass concentrations \((R^2 = 0.99–0.84)\), with depth decay rates \((\lambda)\) consistently higher for mass than numerical concentration. Furthermore, both numerical and mass concentration decay rates were inversely proportional to Beaufort state (Figure 4.3). Depth decay rate of numerical concentration went from 3.0 at Beaufort 1 (95% confidence interval - 95%CI = 2.56-3.45), to 1.7 at Beaufort 3 (95%CI = 1.51-1.88), and 0.8 at Beaufort 4 (95%CI = 0.62-0.98). Decay rate of mass concentration went from 3.8 at Beaufort 1 (95%CI = 3.23-4.33), to 2.4 at Beaufort 3 (95%CI = 1.63-3.14), and 1.7 at Beaufort 4 (95%CI = 1.50-1.94).

These exponential fits had relatively similar depth decay rates to those predicted by Kukulka’s model for Beaufort 3 \((\lambda = 2.36–3.37)\) and 4 \((\lambda = 0.88–1.28)\). However, for Beaufort 1 the statistical fit showed much smaller \(\lambda\) (2.56-4.33) than those predicted by Kukulka’s model \((\lambda = 141.73–47.2492)\).
Figure 4.3 Depth profiles of plastic mass and numerical concentration under different Beaufort scales: 1 (N = 3 net tows), 3 (N = 4 net tows), and 4 (N = 5 net tows).

Black lines show model predictions (Kukulka et al., 2012) using median plastic rise velocity (0.0053 m/s), and the typical range of frictional velocity of water ($u_{\text{friction}}$) at each of the sea states sampled.
4.4.2 Lengths, types, resins and rise velocities of plastics

We counted and classified 12,751 macroscopic plastic pieces with lengths varying from 0.5 to 207 mm (median = 1.5 mm; Figure 4.4). They were mostly fragments of polyethylene (84.7%), followed by polypropylene (15.3%) items. Hard plastics (46.6%) and sheets (45.4%) were predominant, with lower presence of plastic lines (7.9%), pellets (0.05%) and foams (0.008%).

![Figure 4.4 Glass jars with filtered water and plastic samples collected under wind speeds of 1 knot (top image) and 15 knots (bottom image).](image)

From left to right: 0 – 0.5 m, 0.5 – 1 m, 1 – 1.5 m, 1.5 – 2 m, 2 – 2.5 m, 2.5 – 3 m, 3 – 3.5 m, 3.5 – 4 m, 4 – 4.5 m, and 4.5 – 5 m deep.

Plastic rise velocity ranged from 0.001 to 0.0438 m/s (Figure 4.5a). It was directly proportional to plastic length, with the slope of this linear relationship differing among types of plastic (Figure 4.5b). While both hard plastics and sheets had a slope equal to 0.002 (95% CI = 0.0017-0.0026 and 0.0012-0.0023, respectively), plastic lines had a
flatter slope of 0.00007 (95% CI = 0.00002-0.00013), since their rise velocity increased only slightly towards longer pieces. Rise velocities differed among sampled depths, with particles at the surface (0-0.5 m) having a wider range of values and a higher median value than pieces at greater depths (Figure 4.5c).

Figure 4.5 Histogram of rise velocity of plastics (A), plots of plastic sizes x rise velocities of different types of plastic (B), and boxplot of rise velocity at different depth intervals (C).

In C, the central dot is the median value, edges of the box are the 25th and 75th percentiles, whiskers extend to extreme data points not considered outliers, and outliers are plotted individually as crosses.
The vertical mixing process was size-selective, and affected the size distribution of plastics located at the sea surface (Figure 4.6), with the proportion of plastics at depths over 0.5 m generally increasing towards smaller plastic lengths (Figure 4.7). For hard plastics and sheets, this trend was observed at all Beaufort scales sampled. Plastic lines however, only displayed this trend at Beaufort 1, with different size classes showing similar and relatively high underwater proportions at Beaufort 3 and 4.

Datasets produced and analysed in this study are available at Figshare (Reisser et al., 2015).
Figure 4.6 Size histograms of plastics collected at depths 0-0.5 m and 0.5-5 m during Beaufort scale 1 (top panel), 3 (middle panel), and 4 (bottom panel).
Datasets produced and analysed in this study are available at Figshare (Reisser et al., 2015).

4.5 Discussion

This study describes high-resolution depth profiles of plastic concentrations, which were shown to decrease exponentially with depth, with decay rates decreasing towards stronger winds. It also provides the first measurements of the rise velocity of ocean plastics, which varies with particle size and type. Furthermore, it shows that depth profiles of plastic mass are associated with higher decay rates than depth profiles of
plastic numbers. This can be explained by our observation of smaller/lighter plastic pieces generally associated with lower rising velocities, being therefore more susceptible to vertical transport.

Predictions of plastic vertical mixing are commonly used to correct numerical concentrations obtained by surface net sampling (chapter 2) (Kukulka et al., 2012, Cózar et al., 2014, Law et al., 2014). As determined in our study, the model described in Kukulka et al. (2012) performed relatively well in estimating the total number of plastic pieces at the wind-mixed surface layer. The major difference between this model and our observations occurred at the calmest sea state condition (Beaufort scale 1): while the model predicted that all plastics would be at the surface, we still observed some particles submerged at depths greater than 0.5 m below the water surface. This could have been a consequence of the presence of other types of vertical flow at our sampled sites (e.g. downwelling) or the occurrence of plastics rising from deeper waters due to previous wind-driven mixing events.

Our results indicate that plastic numerical concentration decays at a lower rate than plastic mass concentration, as smaller plastics are more susceptible to vertical transport. The uncertainties related to how plastic numerical concentration translates into plastic mass concentration have already led to differences between plastic load estimates arising from different studies. For instance, Cózar et al. (2014) used a correlation based on simultaneous surface tow measurements of total mass and abundance of plastic to convert depth-integrated numerical concentrations into mass concentrations. These authors estimated that the total plastic load in the world’s sea surface layer is between 7,000 and 35,000 tons. On the other hand, Law et al. (2014) multiplied depth-integrated numerical concentrations by the average plastic particle mass (1.36 x 10^{-5} kg), and estimated that the microplastic load at the North Pacific accumulation zone alone is of
at least 21,290 tons. Such differences evidence the importance of better predicting the vertical transport of ocean plastics to develop standard plastic load estimation methods. More sampling is required to better quantify both profiles of plastic mass and numerical concentration over a broader range of sea states, and translate these observations into prediction models. Such models may need to be three-dimensional, and account not only for wind mixing effects, but also ocean plastic properties (e.g. particle size) and other types of vertical transport processes (e.g. Langmuir circulation).

As shown here, and in two modelling studies (Ballent et al., 2012, Isobe et al., 2014), vertical mixing affects the size distribution of plastics floating at the surface. We observed that the proportion of plastics mixed into deeper waters increases towards smaller sizes even under low wind speed (1 knot) conditions (see Figure 7). This observation has implications for studies assessing size distribution of plastics using surface sampling devices. Cózar et al. (2014) and Eriksen et al. (2014) quantified the size distribution of ocean plastics from worldwide sampling locations and concluded that there are major losses of small plastics from the sea surface. Here we show that at least a fraction of this ‘missing’ plastic could be just under the sampled surface layer (0-0.5 m). For instance, 20% of 0.5-1 mm, 13% of 1.5-2 mm, and 8% of 2.5-3 mm long plastics were between 0.5 and 5 m deep during our Beaufort scale 1 net tows. More at-sea and experimental work is required to further quantify this effect and estimate depth-integrated size distribution of buoyant plastics drifting at sea.

Predicting the vertical mixing of buoyant plastics is also important as it affects the horizontal drifting patterns and ecological impacts of plastic pollution. For instance, larger pieces of plastic coming from land-based sources may stay trapped near the shore until further fragmentation, due to a combination of their high buoyancy and the effect of Stokes drift produced by waves parallel to coastlines (Isobe et al., 2014).
Furthermore, the vertical distribution of plastics will influence the likelihood of animals inhabiting different depths to encounter, and potentially interact, with plastic. For instance, sea birds, turtles, and mammals, which breathe air and use the sea surface for daily activities, present high rates of plastic ingestion and entanglement (Derraik, 2002, Tourinho et al., 2010). These high interaction rates could be partly explained by the relatively high concentrations of plastic debris at the sea surface, as shown in this study.

Our findings show that vertical mixing affects the number, mass, and size distribution of buoyant plastics captured by surface nets, a standard equipment for at-sea plastic pollution sampling (Hidalgo-Ruz et al., 2012). Subsurface samples are still scarce and the processes influencing distribution of plastics throughout the ocean’s water column are poorly understood. Further multi-level sampling across a broader range of sea states is necessary for better quantifying the vertical mixing of buoyant plastics. This will improve predictions of ocean plastic concentration levels (Kukulka et al., 2012), size distributions (Cózar et al., 2014, Eriksen et al., 2014), drifting patterns (Isobe et al., 2014), and interactions with neustonic and pelagic species of the world’s oceans.
Chapter 5  General Discussion

Chapters 2, 3, and 4 contain detailed discussions of my PhD results so here I will limit discussion to the key findings related to the overall goals of this thesis, which were to investigate how buoyant plastics are distributed in sea surface waters (both horizontally and vertically), and characterise organisms on the surface of millimetre-sized marine plastics. Furthermore, I will discuss a few limitations of this thesis and suggest some future research directions, which complement those mentioned in the discussion sections of chapters 2, 3, and 4. Finally, the overall conclusions of this thesis are presented.

5.1 Horizontal distribution of buoyant plastics at sea

In chapter 2, I showed that each square kilometre of Australian surface waters is contaminated by thousands of small plastic fragments, mostly smaller than 5 mm across – the so-called “microplastics”. This is the first study to sample plastic debris in waters around Australia and the data collected here comprise most of the very scarce measurements of plastic contamination at surface waters of the Indian Ocean and western South Pacific. This dataset has been made open access in Figshare (Reisser et al., 2013) and is already being used by a few groups of researchers attempting to quantify plastic pollution levels and distribution at the world’s ocean, e.g. (Cózar et al., 2014, Eriksen et al., 2014) (Figure 5.1).
Figure 5.1 Measurements of plastic numerical concentrations used in Cózar et al. (2014).
Gray areas show accumulation zones as predicted by Maximenko et al. (2012). Dots close to the Australian and Fijian continent are data from this thesis. Source: National Geographic
I found that the small plastics in surface waters around Australia are mostly a by-product of the degradation of larger objects made of polyethylene and polypropylene. The high prevalence of small plastic fragments in Australian waters is consistent with other regions of the world’s oceans, where microplastics were found to be the most numerically abundant type of debris in all types of marine environment (Moore et al., 2001, Thompson et al., 2004, Browne et al., 2010, Law et al., 2010, Browne et al., 2011, Eriksen et al., 2013). Plastic pollution levels were moderate when compared to concentrations in other marine areas (Moore et al., 2001, Yamashita and Tanimura, 2007, Law et al., 2010, Collignon et al., 2012, Eriksen et al., 2013), but higher amounts were found close to cities on Australia’s east coast, as well as in remote locations (west Tasmania and North West Shelf). Recent studies reported toxicological effects of these small and contaminated plastics on a host of organisms, from zooplankton, small fish and turtle hatchlings, to large mammals (Basheer et al., 2004, Choy and Drazen, 2013, Cole et al., 2013, Fossi et al., 2012, Gassel et al., 2013, Rochman et al., 2013c, Wright et al., 2013). As such, small plastics are a type of harmful marine debris, implying that plastic hazards to Australian species and ecological communities are likely to be broader than those officially recognized, which only includes physical impacts of large plastic objects on marine vertebrates through entanglement and ingestion (Commonwealth of Australia, 2009).

Additionally, I found that the abundant and widespread small marine plastics around Australia are likely coming from a variety of domestic and international, land- and ocean-based sources. Although marine plastic pollution is a global environmental issue, mostly caused by our massive production of plastic single-use disposable items, there are still no attempts to regulate plastic disposal on land and directly at sea at an international level (Rochman et al., 2013a). Further at-sea studies on the
characterization, spatial distribution, and pathways of marine plastics in coastal and oceanic regions around Australia, as well as on marine plastic toxin loads and interactions between small plastic particles and organisms at all trophic levels of the food web, are necessary.

Even though the pioneering study described in chapter 2 advanced our knowledge about plastic pollution in Australian waters, it is important to recognise the limitations of this opportunistic research aboard Australian vessels. Among them are (1) the relatively small number of surface trawls conducted, resulting in a very descriptive study; (2) the use of two types of surface samplers (i.e. Manta and Neuston nets), which may have brought some sampling bias to the plastic concentration values obtained; (3) the estimation of plastic numerical concentrations only, as weighing the samples would have damaged the plastic particles, impeding the conduction of the SEM and FT-IR analyses described in chapter 3; (4) the use of a mean rising speed different from the one I obtained two years later, using plastics from the North Atlantic accumulation zone (chapter 4); (5) the lack of wind measurements in two out of the seven voyages conducted, which did not allow the use of in-situ wind measurements to calculate the depth-integrated plastic concentrations; and (6) the quantification of amounts of debris within a limited size range (i.e. zooplankton size range only), as Manta trawl surveys are not ideal for the quantification of microscopic plastic and mega debris.

I suggest that researchers continue monitoring plastic pollution levels in Australian waters and beyond, by taking advantage of transit voyages aboard research vessels (e.g. RV Investigator, Solander, Falkor). “Underway” data related to plastic debris could include surface net sampling, as well as visual transects for counting larger items of marine plastic debris (Eriksen et al., 2014). Furthermore, simple methods using continuous seawater intake of vessels (Lusher et al., 2014) could be applied during a
wider range of maritime operations, thus maximising sea time and sampling effort. Given the limitations of opportunistic sampling, I also suggest the planning and execution of dedicated voyages to study plastic pollution. These voyages would provide the plastic pollution research community with the opportunity to conduct robust surveys with well-planned sampling designs. Plastic pollution research voyages with extensive sea time, fund, and human resources, could use a range of aerial, sea surface and underwater surveys to quantify concentrations of plastic debris within a broad range of debris sizes, from nanometre-sized microplastics to meter-sized ghost nets.

5.2 Organisms on the surface of millimetre-sized ocean plastics

Chapter 3 showed that millimetre-sized plastics contaminating Australian waters are home of a host of marine life (Figure 5.2), which is being transported by ‘plastic drifting’, potentially affecting the fate and environmental impacts of plastic pollution. It contributed 18 new records of taxa living on microplastics, and provided 1,143 open access high-resolution images of plastic debris shapes, surface textures, and ‘epiplastic’ communities (Reisser et al., 2014).
Figure 5.2 False-coloured electron micrographs of surface textures and organisms on buoyant plastic debris. Original images (i.e. black and white) are available in Reisser et al. (2014a)

Top left: diatoms (green) and rounded cells (purple). Top right: Coccoliths (red, blue, yellow) and diatoms (green, turquoise). Bottom left: unidentified marine worm (red). Bottom right: isopod (red). Source: The Conversation

This study evidenced that microscopic plastic-dwellers are everywhere in our oceans. Organisms ranging from single-celled microbes to invertebrates are all taking advantage of this new human-made and durable type of floating habitat. Even though the SEM work described here advanced our knowledge on the types of microbes inhabiting plastic debris, this visual technique has many limitations, such as the inability to identify the vast majority of microbes composing the ‘Plastisphere’ (e.g. bacteria and fungi species). A more detailed study of epiplastic organisms could be achieved by analysing ocean plastic biofouling with other modern techniques, such as metagenomics (Thomas et al., 2012) and fluorescence in situ hybridization (Moter and Göbel, 2000).
Interesting observations of the SEM work described in this thesis included the many flourishing microbes that appeared to interact with the plastic surfaces. These observations, together with findings from previous studies (Harrison et al., 2011, Zettler et al., 2013), suggest that microbes are helping break down plastics at sea. These putative hydrocarbon-degrading microbes might support biotechnological solutions for better plastic waste disposal practices on land (Sivan, 2011). For instance, such marine microbes or their genes might enable the development of industrial ‘composts’ able to break down plastic waste on land, thus reducing the risk of plastic findings its way to the sea.

A shipboard three-dimensional survey to collect ocean plastics for functional genome studies could significantly enhance our understanding of the plastic biofilm in a relatively quick manner. Microplastics collected at the sea surface, water column, and seafloor could have their biofilms screened for hydrocarbon-degrading genes, and results compared with those obtained from the biofilm of natural substrates, such as sediments, pumice, drifting wood, and floating seaweed. The detection of relatively high amounts of hydrocarbon-degrading genes in the ‘Plastisphere’ may be the first step towards the development of a bioremediation solution to some of our plastic waste management issues.

5.2.1 Ingestion of plastics at sea: does debris size really matter?

Marine microplastics (< 5 mm in length) can contain high loads of additives and adsorbed pollutants, and may be a threat to marine food webs due to their ingestion by organisms at the base of the food chain (http://www.unep.org/yearbook/). Most of our knowledge of plastic ingestion by zooplankton has been obtained through experiments assuming that plastic particles have to be smaller than the organism’s feeding apparatus
for this type of interaction to occur (Cole et al., 2013). However, I propose that this is not a rule.

By examining the surface of millimetre-sized marine plastics using a scanning electron microscope, I observed a diverse range of fouling organisms, and a variety of intriguing pits and scraping marks of unknown origin - see details in chapter 3 and SEM images at (Reisser et al., 2014). I suggest that some of these plastic surface textures are feeding marks produced by invertebrates grazing upon the plastic biofilm.

I observed sub-parallel linear scrapes with spacing of 5-14 μm (Figure 5.3a,c), which is similar to typical distances between teeth of the mandibular gnathobases of copepods (Michels et al., 2012). The thinner and shallower marks around the linear scrapes could have been formed by filamentous microstructures present on their gnathobases.

Copepods are an abundant planktivorous group and possess strong feeding apparatuses to feed upon organisms such as diatoms with silica cell walls (Michels et al., 2012). Some pelagic species have flexible feeding habits, and can feed on sea-ice algae (Brierley and Thomas, 2002), faecal pellets (Gonzalez and Smetacek, 1994, Noji et al., 1991), and marine snow particles (Turner, 2002). I suggest that these copepods could also feed upon biofilm of plastic debris, which is often rich in ‘epiplastic’ diatoms; see chapter 3 and (Carson et al., 2013).
Figure 5.3 Scrapes putatively identified as feeding marks.

a: Linear scrape marks on a 2.3 mm long plastic debris with a high load of diatoms. b: Rounded scrape marks on a 6 mm long plastic with an unidentified marine worm. Arrow indicates unknown structure partially covering the worm. c: zoom on scraping displayed in ‘a’. d: zoom on scraping shown in ‘b’. Scale bars = 10 μm (a, c), 100 μm (b), 20 μm (d)

I also observed peculiar, rounded marks close to an unidentified marine worm (Figure 5.3b,d), which was partially covered by an unknown structure (indicated by the arrow) possibly secreted by the animal. These unique scraping marks were also noted on two other plastic pieces that did not have any visible animals, but possessed structures similar to the one covering the worm in Figure 5.3b. These results suggest that feeding on plastic biofilm is not restricted to zooplankton, and possibly occurs with rafting organisms such as amphipods, gastropods, and chitons, which are known to associate with floating debris such as plastics (Winston et al., 1997).

Small portions of the plastic particles were apparently removed, and perhaps ingested, during these putative grazing activities (Figure 5.3). Thus, I contend that (1) plastic
biofouling induces plastic ingestion, and (2) plastic pieces must not necessarily be smaller than the organism for a feeding interaction to occur. The latter hypothesis has already been suggested for large items, as 15.8% of drifting plastic objects in Hawaii displayed a variety of vertebrate bite marks (Carson, 2013).

Experiments exposing zooplanktonic organisms to millimetre-sized plastics with biofilm may document whether they are capable of handling these particles, creating such feeding marks. By exposing neustonic zooplankton to fresh pieces of brittle plastic debris, researchers could possibly document this new type of feeding behaviour (e.g. by filming) and detect plastic bits co-ingested with biofilm grazing (e.g. by examining faecal pellets).

Due to their rapid growth and nutritional value, biofilms on plastic debris may be a significant new food source for invertebrates, particularly in the oligotrophic waters within subtropical gyres, where plastic contamination levels are particularly high. The impacts related to this new type of feeding interaction remain unclear, but are likely negative since plastics pose chemical and physical threats to their ‘predators/grazers’ (Wright et al., 2013). These impacts could include effects on food webs, since plastic-associated pollutants and additives could be transferred to the biofilm and moved up the food chain of plastic ‘predators/grazers’. The implications of plastic biofilm ingestion, particularly in terms of pollutant transfer and health effects, should also be investigated.

5.3 Vertical distribution of buoyant plastics at sea

Chapter 4 is the first study to acquire high-resolution depth profiles of buoyant plastic concentrations and characteristics across a range of sea states. This study described high-resolution depth profiles of plastic concentrations, which decreased exponentially with depth, and provided the first measurements of the rise velocity of ocean plastics,
which varied with particle size and type. Furthermore, it showed that depth profiles of plastic mass are associated with higher decay rates than depth profiles of plastic numbers. This can be explained by the observation that smaller/lighter plastic pieces were generally associated with lower rising velocities, being therefore more susceptible to vertical transport.

My findings show that vertical mixing affects the number, mass, and size distribution of buoyant plastics captured by surface nets, a standard equipment for at-sea plastic pollution sampling (Hidalgo-Ruz et al., 2012). Subsurface samples are still scarce and the processes influencing distribution of plastics throughout the ocean’s water column are poorly understood. Further multi-level sampling across a broader range of sea states is necessary for better quantifying the vertical mixing of buoyant plastics. This will improve predictions of ocean plastic concentration levels (Kukulka et al., 2012), size distributions (Cózar et al., 2014, Eriksen et al., 2014), drifting patterns (Isobe et al., 2014), and interactions with neustonic and pelagic species of the world’s oceans.

Models capable of using sea surface measurements of plastic sizes and concentrations to predict depth-integrated values are of extreme importance to those attempting to quantify and characterise marine plastic pollution. Kukulka et al. (2012) developed a simple model to predict depth-integrated numerical concentration (pieces km\(^{-2}\)) of plastics. However, predictive models of depth-integrated mass concentrations and size distribution still need to be developed. These depth-integrated values would then be comparable without underestimating plastic loads and amounts of small microplastics within the mixed layer. More multi-level observations such as the ones described in this thesis, both in oceanic and coastal waters, will facilitate the creation of such models.

There is an urgent need to develop three-dimensional surveys capable of detecting
plastic debris thought the water column. Perhaps the unique optical properties of plastics in the infrared band (e.g. see Figure 2.3) could be used to develop a sensor capable of detecting ocean plastics, thus making plastic contamination quantification more efficient and independent of human observers. Since hydrocarbon-bearing substances such as plastics have typical absorption maxima around 1730nm and 2310 nm (Höring et al., 2001, Tsuchida et al., 2009), short-wave infrared and Raman spectroscopy hold promise for such application.

A sampling equipment that may improve the way we sample microplastics at-sea is the Video Plankton Recorder (Davis et al., 1992), which is an underwater microscope capable of capturing images of microorganisms and particles while being towed by a vessel. This equipment could provide valuable information on the horizontal and vertical distribution of small plastic debris.

5.4 Overall Conclusions

The ocean surface is heavily contaminated by plastic pollution, mostly in the form of fragments coming from the degradation of plastic objects. The most common types of items lost or discarded at sea are packaging and fishing gear made of polyethylene and polypropylene. These are two very common types of plastic resin that are lighter than seawater and quite resistant to environmental degradation. Such characteristics make them particularly prone to long-distance dispersion in the ocean surface. Buoyant plastic represent a major hazard to organisms that live in the top layer of the oceans (e.g. sea turtles, birds, neuston communities); they are also a new type of vehicle for the dispersal of pollutants and organisms that live associated with these synthetic floating habitats.
This study set out to explore the spatial distribution and impacts of ocean plastic pollution. By performing net tows for sampling surface plastics, I was able to characterize the plastic debris in surface waters around Australia, describe textures and organisms on the surface of millimetre-sized ocean plastic, and quantify the way buoyant plastic is distributed in the top layer of the oceans.

The thesis showed that surface waters around Australia have moderate contamination levels of microplastics, which were mostly fragments of larger objects made of polyethylene and polypropylene. These millimetre-sized particles harbour a large variety of organisms, such as bacteria, diatoms, and invertebrates, and are concentrated at the sea surface, although they can reach deeper depths depending on their buoyancy and sea state. The information presented in this thesis greatly contributed towards a better understanding of the distribution and potential impacts of plastic pollution in the world’s oceans.

It is important that continued research be conducted to further improve this baseline knowledge for defining regulations and mitigation strategies for this concerning issue. The findings of this thesis highlight the need for more plastic pollution research, monitoring, and mitigation efforts within Australian waters, as well as a better quantification of the depth profile of buoyant plastic debris. This would greatly improve our capacity to quantify, characterize and search for solutions related to this new type of marine pollution.

To generate achievable policy strategies and develop targets with regards to plastic pollution reduction, there is need for more research and monitoring projects at the local, national, and international scales aiming at assessing the environmental, economical and human health impacts of plastics. Furthermore, collaborative research and development
efforts towards better environmental awareness, waste management, packaging designs, bioplastics, cleanup efforts, and regulation enforcements will help mitigate and solve this growing environmental issue.
References


PLASTIC WASTE MANAGEMENT INSTITUTE. 2013. *Plastic products, plastic waste and resource recovery* - - - [Online].


Appendix 1 Outputs produced during this candidature

Peer-reviewed publications published, in press or in prep


Media articles


Appendix 2 Chapter 2 supplementary material

Net tow data (N = 171) Columns indicate net station number, sampling date (day.month.year), location (degrees minutes), and mean sea surface plastic concentration (Cs; pieces per km²).

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Probability distribution of virtual particles arriving at the 57 net stations of this study (dispersal time = 45 days). Red dots indicate position of net stations.

**June 2011**

**August 2011**

**April 2012**

**May 2012**
**Real drifter pathways arriving at the 57 net stations.** Purple dots indicate net station locations and asterisks indicate drifter release areas.
Appendix 3 Chapter 3 supplementary material

Examples of marine plastic surface textures. *a, d*: polypropylene plastics with linear fractures and pits; *b, c*: higher magnification of the plastic surface shown in ‘*a*’ (note very similar pits – one empty and one with a cell conforming its shape); *e*: higher magnification of the plastic surface shown in ‘*d*’ (note three equally spaced deep pits); *f*: polyethylene soft plastic with linear fractures, producing squared microplastics; *g*: higher magnification of the plastic surface shown in ‘*f*’ (note shallow pits likely formed by *Cocconeis* sp.); *h*: rounded scrape mark similar to the ones found close to the worm-like animal (see Figure 3.7i); *i, k*: sub-parallel scrape marks; *j*: large plastic pit likely formed by an egg of *Halobates* sp.