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Using hydrodynamic models to understand the impacts and risks of plastic pollution

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Wildermann N, **Critchell K**, Fuentes MMPB, Limpus CJ, Wolanski E, Hamann M. Does behaviour affect the dispersal of flatback post-hatchlings in the Great Barrier Reef? *Royal Society Open Science*. 2017; 4(5).

Critchell K, Grech A, Schlaefer J, Andutta FP, Lambrechts J, Wolanski E, modelling the fate of marine debris along a complex shoreline: lessons from the Great Barrier Reef. *Estuarine, Coastal and Shelf Science*. 2015; 167:414-26.

Conference presentations

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Critchell K, Lambrechts J, Wolanski E. 2015. 'Modelling the fate of plastic debris in coastal waters' *Estuarine and Coastal Sciences Association Conference*, London, UK

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Abstract

Anthropogenic marine debris, mainly of plastic origin, is accumulating in estuarine and coastal environments around the world, causing damage to multiple species of fauna and flora, as well as habitats. Plastics have the potential to accumulate in food webs, and cause economic losses to tourism and sea-going industries, like commercial fishing. The production and use of plastic products is growing, from 230 million tonnes produced globally in 2005 to 320 million tonnes in 2015, a 40% increase in production over 10 years. If we are to manage the increasing input and threat, we must understand where plastic pollution is accumulating in the environment and what the impacts to organisms in these areas are.

The goal of this thesis was to explore the dispersal and risks of plastic pollution in the coastal environment, at a scale that is useful to local management authorities. I used four research aims to achieve this goal. The aim of the first data chapter (Chapter 2) was to prioritise research that would improve modelling outputs in the future. In the second data chapter (Chapter 3), the aim was to locate the areas of highest exposure to plastic pollution for three vulnerable habitats. In the third data chapter (Chapter 4), I aimed to explore the dominant sources and processes of plastic accumulation. Lastly, in the final data chapter (Chapter 5), I aimed to understand the sub-lethal consequence of plastic exposure on a tropic reef fish.

The first data chapter of my thesis presents an advection-diffusion model that includes beaching, settling, resuspension/re-floating, degradation and topographic effects on the wind in nearshore waters to quantify the relative importance of these physical processes in governing plastic debris accumulation. I found that the source location has by far the largest effect on the accumulation location of the debris. The diffusivity, used to parameterise the sub-grid scale movements, and the relationship between debris resuspension/re-floating from beaches and the presence of a wind shadow created by high islands also has a dramatic impact on the modelled accumulation areas. The rate of degradation of macroplastics into microplastics also had a large influence in the prediction of debris dispersal and accumulation. These findings may help prioritise research on the physical processes that affect plastic accumulation, leading to more accurate modelling, and subsequently an improved empirical basis for management in the future.

In the second data chapter, I used the model described in Chapter 2 to predict the potential exposure of vulnerable habitats and species to plastic pollution using a spatial risk assessment approach. The effect of plastics on the marine environment is well documented, however the physical location of these interactions are largely unknown. I assessed the potential exposure of

mangroves, coral reefs and marine turtles to plastics during the two main wind conditions of the region; the trade winds and monsoon wind seasons. By creating relative exposure categories based on the density of particles in modelling outputs of nil, low, medium and high exposure. I found that in the trade wind season (April to September, dominated by strong south-easterly winds) marine turtles, mangroves and reef habitats had lower exposure than during the monsoon wind season (October to March, dominated by lighter and more variable winds). A small proportion of coral reef habitat was in the high exposure categories, whereas the turtle home-range had a large area in high exposure categories (16% and 26% exposed to high microplastics during monsoon season, respectively). Unlike the other two case studies, the mangrove habitat had consistent hotspots of high exposure across both wind seasons. The outputs of this chapter can inform local scale management action, for example turtle management and recovery plans. The method presented here can also be transferred to other species and habitats and scaled up for larger jurisdictions.

In the third data chapter, I built on Chapter 2 (a sensitivity analysis of physical/modelled processes) by using field data for macro- and microplastics to interrogate the model. The aim was to find the likely sources of plastics to the Whitsunday region and understand the limitations for the model in a complex coastline and at a management-relevant scale. I found that, for microplastics, offshore sources are likely to be more important than onshore, and for macroplastics, local (onshore) sources are more important than they are for microplastics. Of the physical characteristics I examined, I found none that make a site more or less predictable in the modelling. Field data on sources at local scales is necessary, although, this is recognised as a difficult task.

In the last data chapter, I assessed the consequence of plastic exposure by quantifying the effect of microplastic exposure on juveniles of a widespread and abundant planktivorous fish (*Acanthochromis polyacanthus*). Under five different plastic concentrations, with plastics the same size as the natural food particles (mean 2 mm diameter), consumption of microplastic was low and there was no significant effect of plastic exposure on fish growth, body condition or behaviour. However, the number of plastics found in the gut of the fish vastly increased when plastic particle size was reduced to approximately one quarter the size of the normal food particles, with a maximum of 2102 small (< 300 µm diameter) particles present in the gut of individual fish after a 1-week plastic exposure period. Under conditions where food was replaced by plastic, there was a negative effect on the growth and body condition of the fish. These results suggest plastics could become more of a problem as they breakup into smaller size classes, and that environmental changes that lead to a decrease in plankton concentrations likely have a greater influence on fish populations than microplastic presence alone.

The risks of plastic pollution to environmental features remain largely unquantified. However, my thesis demonstrates significant gains in understanding of mechanisms that can be used to determine where plastics are likely to accumulate, and identifies priorities for future research to improve the statistical power of the models. For example, by understanding the resuspension of plastic in areas without wind driven waves. This thesis also highlights the need to understand different types of plastics separately, microplastics have different consequences to macroplastics, different areas of accumulation and different sources, therefore the risks and appropriate management actions are very different.

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Statement of contributions

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Publication plan with contributions of co-authors

Chapter Number	Publication details	Extent of the intellectual input of each author, including the candidate
2	Critchell K, Lambrechts J. Modelling accumulation of marine plastics in the coastal zone; what are the dominant physical processes? *Estuarine, Coastal and Shelf Science*. 2016; 171:111-22.	I developed the research question. Lambrechts and I collected the data. I performed data analysis. I wrote the paper with input from editorial Lambrechts. I developed the figures and tables.
3	Critchell K, Hamann M, Wildermann N, Grech A. A spatially explicit exposure analysis of plastic pollution in the Whitsunday region *in prep*	I developed the research question. Wildermann and I collected the data. I performed data analysis under the guidance of Grech. Wildermann conducted home range analysis. I wrote the paper with editorial input from all. I developed the figures and tables under the guidance of Grech.
4	Critchell K, Hoogenboom M, Grech A, Wolanski E, Hamann M. Using field data to interrogate a plastics dispersal model *in prep*	I developed the research question. I collected the data. I performed data analysis under the guidance of Wolanski, Hamann, Grech and Hoogenboom. I wrote the paper with editorial input from all. I developed the figures and tables.
5	Critchell K, Hoogenboom M. Effects of microplastic exposure on the body condition and behaviour of planktivorous reef fish (*Acanthochromis polyacanthus*) u*nder review*	Hoogenboom and I developed the research question. I conducted the experiment and data collection under the guidance of Hoogenboom. I performed the data analysis under the guidance of Hoogenboom. I wrote the paper with editorial input from Hoogenboom. I developed the figures and tables with support from Hoogenboom.

Permit approvals and ethics statement

All necessary permits and approvals to conduct this work. Fieldwork was undertaken under the Great Barrier Reef Marine Park and National Parks joint permit number G15/37509.1.

The fish experiment was carried out in strict accordance with Ethics protocol laid out by the *Animal Ethics Committee of James Cook University,* who approved the protocol of these experiments (Permit number: A2112). Priority was given to animal care at all stages of this study.

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Chapter 1
General Introduction

Over recent decades, the increase in population, technological developments and urbanisation have increased pollution in the environment. Some of the earliest pollutants were coal ash, industrial waste and sewage in the mid-19th century, coinciding with the industrial revolution. It was not until the 1950s that the environmental movement began taking action towards reducing pollution inputs with legislation for clean air the America and the UK (Air Pollution Control Act, 1955; Clean Air Act, 1956, respectively), among others. The use of many substances deemed to contribute to pollution is controlled by legislation or other policy instruments aiming to prevent or minimise environmental harm. The management of point source and diffuse source pollutants differ. For point source pollutants, the source is known and identifiable, therefore legislating to set maximum emission limits is relatively common. For example, limits are set to manage emissions of heavy metals or persistent organic pollutants (POPs) by various industrial sectors. Diffuse source pollutants are more difficult to legislate for, essentially because it is not possible to locate the source, or the use is wide-spread and thus there is no single source contributing to the pollution. Dichlorodiphenyltrichloroethane (DDT), for example, is a chemical which has been used extensively as an agricultural insecticide for many decades, and although it is still used in some parts of the world as a malaria control method, its use is illegal in most developed nations due to its impact on the environment (UNEP, 2002; van den Berg, 2009). Despite these regulations across the globe, our air, soil and water systems are all experiencing an increased presence of pollutants.

Plastic is a pollutant of emerging concern and notoriety in the environment. Plastic-based products occur in many forms and are widely used around the world. One of the key reasons is that plastic forms the base material for a growing number of products used in society, and it has a large, and growing, number of applications. Some of these uses for plastic have revolutionised industries, for instance, modern medicine has benefited from plastic materials such as those used to make flexible, durable equipment or the sterile single-use equipment to minimise infection (Naik, 2017). In agriculture, plastic ground sheeting used for some crops has led to reduced use of herbicides and the amount of water necessary to produce the crop (Steinmetz et al., 2016), and this has improved runoff from agricultural land into coastal systems and profit margins of the industry. The use of plastics in automobile and airline industries has led to lighter vehicles or loads thus increasing the cost-efficiency of travel and transport. However, it is the low cost and ease of availability of plastics that has led to their use in society as convenient items, or packaging, designed to be disposable. Such is the global demand for plastics that in 2015 alone, 322 million tonnes were produced, a 40% increase in a decade from 230 million tonnes in 2005 (Plastics

Europe, 2016). Of this, it is estimated that a third of the plastic production goes to single-use packaging products.

With the increase in production and dependency of plastic-based materials, it is inevitable that plastic will end up in the natural environment. Plastic products can end up in the environment through irresponsible disposal, such as littering, accidental leakage from land-fill, and inadvertent loss through municipal waste-water treatment. Streets and storm drains often flow directly into freshwater systems where the plastics they carry accumulate (Eerkes-Medrano et al., 2015; Anderson et al., 2016; Cable et al., 2017). From the freshwater systems, the plastics are often washed into coastal and oceanic environments and these have become some of the most infamous accumulation zones of plastics in the environment. Plastics can also be deposited directly into the ocean though dumping at sea from vessels or from lost or otherwise discarded fishing gear. It is impossible to know with certainty how much plastic is in the ocean, however, estimates range from 5.2 trillion pieces (Eriksen et al., 2014) to 15 to 51 trillion pieces (van Sebille et al., 2015). Wherever the true value lies, the plastic load is large and increasing. Indeed, Jambeck et al., (2015) estimate between 4-12 million tonnes of plastics enter the ocean from land-based sources every year. Australia contributes a relatively small amount to the global problem due to the small population (Jambeck et al., 2015), however, there are significant accumulation areas on beaches all around the coast (Hardesty et al., 2017).

Plastics in the environment break-up into smaller and smaller pieces primarily because the polymer bonds undergo photo-degradation in the presence of Ultra-Violet (UV) light. In the ocean, decreased temperatures, and UV light attenuation, increase the time for plastics to degrade when compared to on land. Degradation on intertidal/exposed areas can be quite rapid. Weinstein et al., (2016) found that fragmentation of high-density polyethylene, polypropylene, and extruded polystyrene started at eight weeks in field experiments at Charleston, SC, USA (32^0 North). As plastics break-up into smaller pieces we classify them into size classes. Microplastics are generally regarded to be those smaller than 5 mm but larger than 100 µm, and macroplastics that are those larger than 5 mm. These size definitions are most common, however, they are not the only definitions that have been used in the scientific literature (Andrady, 2011). Plastics can enter the marine environment as macro- or microplastics, but the most commonly recognised source of microplastics comes from degraded macroplastic (Andrady, 2011).

The scale of the marine plastic pollution issue is global. Most of the present load is derived from urban areas (Schmidt et al., 2017) and, once plastics are in the marine environment, the combination of currents and wind move particles across seas and oceans for years (Ebbesmeyer et al., 2007; Maximenko et al., 2015). Indeed, some of the most publicised accumulation zones are the ocean gyres, generally located close to the centre of each ocean basin and driven by large-scale currents (Lebreton et al., 2012). The high degree of dispersal also makes the quantification and removal of plastics particularly difficult, because the plastics can disperse great distances away from their source, and accumulation zones are often a considerable distance from land. Furthermore, plastic products take a long time to break-up completely, but through time the object loses identifying marks and features, and as it remains in the environment for long periods of time, completing many circulations of the ocean, the sources are difficult to determine.

The occurrence of negative interactions between plastics and the environment has been well documented (Derraik, 2002; Andrady, 2011; Wright et al., 2013b). There have been many studies describing the observations of negative interactions to species with plastic pollution (see reviews by Thompson et al., 2009, Andrady 2011, and Chae and An 2017). The interactions with organisms, especially marine megafauna, are especially well documented (Baulch and Perry, 2014; Nelms et al., 2016). Ingestion and entanglement are the most widely recognised threats to aquatic animals. Entanglement in lost or otherwise discarded fishing gear, rope and plastic sheeting can reduce the ability of the animal to feed, move and behave normally (Gregory, 2009; Vegter et al., 2014; Duncan et al., 2017). It is also widely recognised that entanglement can damage the animal's flesh or amputate/decapitate through blood restriction (Gregory, 2009). Entanglement can also, in the case of air-breathing aquatic animals, cause the animal to drown by weighing it down, reducing the animal's ability to swim to the surface (Vegter et al., 2014; Nelms et al., 2016; Duncan et al., 2017). Ingestion of plastic material can cause damage to the digestive tract (Parga, 2012; Baulch and Perry, 2014) through lesions, abrasions and digestive blockages. If plastic particles are ingested and not passed through, they can cause physical blockage of the tract, as the particles fill the stomach causing starvation by reducing the feeding stimulus and reducing stomach capacity (Ryan, 1988). Ingested plastics can also transfer toxic substances to the animal causing disruption of the endocrine system (Rochman et al., 2013; 2014). Plastic pollution can also cause damage to habitats through smothering, scouring (e.g. Uneputty and Evans, 1997; Donohue et al., 2001) and changed physical properties (e.g. sand permeability and thermal properties associated with microplastic accumulation (Carson et al., 2011). Buoyant plastic objects also act as a vector for invasive species as they can remain a float for far longer than natural objects, allowing infauna and epifauna to travel greater distances (Chae and An, 2017). These impacts to the environment clearly need addressing though management action.

Managers of the aquatic environments (rivers, estuaries, and seas) must balance the allocation of resources to various projects and issues. If management authorities are to use scientific data to underpin initiatives to minimise environmental harm, the science needs to be undertaken at a scale relevant to their management jurisdiction. Essentially, managers require robust data to be relevant to the jurisdictional area and of adequate spatial and temporal resolution to enable action (Fleishman et al., 2011). For example, Sherman and van Sebille (2016) used modelling to find optimal equipment placement locations for microplastic removal at sea. Large-scale projects are useful, however, management intervention is most feasible at the local jurisdictional scale and thus there is a need to conduct data collection and analysis at these smaller, jurisdictionally relevant, spatial scales. In Australia, local councils (municipal) and state government agencies are responsible for waste management, litter prevention and clean-up activities. The benefit of local management (council) intervention is that plastics can be prevented from entering the marine ecosystems at the source (Willis et al., 2017). Prevention of input at the source is cheaper, more effective, and ultimately reduces the subsequent problems, which can’t be easily resolved (Eagle et al., 2016; Willis et al., 2017). Knowing which individual sources to target is difficult without some estimation of which sources contribute the most to the system and where the plastics go after being dispersed from those sources. On a larger, global scale this has been quantified, and Schmidt et al., (2017) estimate that 88-95% of global plastic pollution is entering the ocean from just 10 urban river systems. However, there have been no attempts to quantify this at a scale relevant to local management.

There are many knowledge gaps in the information needed to conduct robust spatially explicit assessments of plastic distribution at spatial scales relevant to management activities. The scale of management changes with each level of government, with the state having a much larger jurisdiction than local councils. On average, local government in coastal regions (local councils) in Queensland, Australia, are responsible for 366 km of coastline. Models of plastic distribution available in the literature (e.g. Yoon et al., 2010; Maximenko et al., 2012) provide approximately seven cells of data across the Queensland coastline, which is therefore inadequate for decision making at scales of local councils. Therefore, this thesis aims to provide data at a scale relevant to local council management area. To achieve this, the spatial scale of the model I use must represent local geographic features in a meaningful way and provide an accurate representation of the hydrodynamics of the area.

1.1 Risk assessment to inform management of plastic pollution

Risk or threat assessment is a widely-used tool that facilitates development of management actions that target complex problems. Risk assessments inform management decisions, weighing up the likelihood and consequence of activities, events and actions (Bottrill et al., 2008). Risk assessments incorporate two quantifiable aspects: likelihood of exposure and consequence of exposure. Risk assessments can be general, where the risk is assessed broadly and the approach results in one value for the risk of the threat. This can be relevant for threats that do not have a spatial component, for example the probabilistic environmental risk assessment of nanomaterials presented by Coll et al., (2016), or those associated with risks to human safety in industrial settings. However, for threats that change concentration or abundance in space and time, for example changing concentration of a pollutant via dilution or the changing density of receptor organisms, this approach is less valid. In these cases, a spatial-based risk assessment is one mechanism that can be used, which provide a map of continuous risk values across a spatially explicit area allowing managers to make spatial-based decisions or designate priority areas for management action (Lahr and Kooistra, 2010). On a global scale, Halpern et al., (2008) assessed the cumulative impacts of anthropogenic activities (stressors) on the marine environment, showing that every area of ocean is impacted by at least one stressor, but despite this global exposure there are areas of relatively low impact. These data allow managers in particular areas to prioritise activities accordingly. However, the data could only be used at the jurisdictional scale of the assessment. To act at a state or local level this resolution would be inadequate.

The exposure of the threat is broken down into the distribution of the threat and the interaction rate with the chosen receptor, such as a habitat or species of animal. The distribution of the threat is often measured using field data and the values between observations are interpolated, or the distribution is modelled based on predictor variables. For example, Srivastava et al., (2012) used water quality field measures in a broader assessment of water quality in India. For pollutants that have fairly homogeneous distributions, or have transport mechanisms that are understood, this method works well. However, for a patchy, variable, and long-lived pollutant in the coastal environment, such as plastics (Barrows et al., 2016; Underwood et al., 2017), this method may be relevant for only the time stamp of field data collection. Collecting samples for marine plastics (especially microplastics) is time consuming and the processing of the samples can take considerable time. Therefore, field data for mapping the distribution of plastic-based pollution is often not feasible at the spatial resolution required for management, especially in areas removed from urban, coastal environments.

The interaction rate is the second component of the exposure parameter of the risk assessment. This is where, and how often, the vulnerable habitat or species (receptor) will encounter the threat. For most threats, including plastic pollution, the rate of interaction with threatened species and habitats is often unknown. The interaction rate is a combination of plastic abundance data along with data on the habitats and the species abundance in the area. Indeed, Nelms et al., (2016) list the need to develop risk maps for sea turtles to be a key future research direction, citing interaction rates as a knowledge gap and suggesting use of oceanographic and niche models to improve knowledge. As an alternative method, Darmon et al., (2017) calculate the interaction rate between turtles and macroplastics in the Mediterranean using aerial survey techniques. They counted the amount of plastics found within a 2 km radius of a turtle to calculate the frequency of interactions. However, this technique would almost certainly be biased towards larger plastic items.

Understanding the consequences of pollutant-organism interactions is the final component of risk assessment. Consequence can be assessed in a few different ways (Lahr and Kooistra, 2010). For example, if concentration effects are known, the probability of an effect can be calculated. Lan et al., (2015) describe a framework for assessing the risk of oil spills that incorporates the impact of the oil and resilience. More simply, if a threshold value is known (e.g. LD_{50}) this can be used to incorporate consequence into spatial risk assessments (Lahr and Kooistra, 2010). Many environmental risk assessments use implied consequence, in that the presence of the pollutant is used to indicate an impact. For example, Fox et al., (2016) use seabird density and oil spill prevalence to assess the risk of oil spills to sea birds on the Pacific Coast of Canada. However, without the consequence component it is likely that the risk is over-stated (Valdor et al., 2017).

There are many factors limiting our understanding of where plastic-species interactions take place that hinder implementation of spatial risk assessments. Firstly, we do not fully understand where the plastics are, at local scales. Field-collected data on plastic concentration is sparse and there is a lack of rigorous spatially-explicit datasets. The available data are often sporadically collected and there is often a lack of unified methods, making comparisons of datasets complicated. These issues occur because the quantification of plastics in the environment is relatively new, and many of the projects around the world are set up to remove plastic waste and are thus not based on robust experimental or field-based sampling designs. We can use sparse field data to inform modelling of concentration predictions, however, the current published predictions are at a coarse spatial scale and not suitable for local management action. In contrast, for many marine receptor organisms we have a relatively good understanding of the spatial distributions, key habitats and population trends. Therefore, for plastic pollution, our estimates of the likelihood of exposure are limited because of our lack of knowledge of plastic pollution sources, dispersal and distribution, although consequence data of also lacking.

The knowledge and understanding of the consequence of plastic interactions with ecological features is increasing. Indeed, a search in the Web of Science for publications with the topics “plastic” and “impact” shows that publication volume has increased from 356 in the year 2000 to 1587 in 2017 (also see Vegter et al., 2014; Nelms et al., 2016). Plastic particles of various sizes in the ocean or waterway expose animals to the threat of entanglement or ingestion. However, there is a lack of data on consequence that takes into account concentration effects of plastics on vulnerable species and habitats, especially in tropical regions. The number of species known to be impacted by plastic is large (Derraik, 2002; Andrady, 2011; Chae and An, 2017), and well documented in comparison to what is known about specific habitats, such as mangroves, seagrass, and coral reefs which have very limited data on the effect of the exposure to plastics. These knowledge gaps mean we don’t really understand risk of plastic pollution in the environment, especially with respect to its spatial accumulation.

There are only a few examples of spatially explicit risk assessments for plastics in the literature. Wilcox et al., (2013), present a study of the risk of entanglement to sea turtles by ghost nets in the Gulf of Carpentaria. They use a model of ghost net distribution derived from drifter buoys to estimate most likely areas of marine turtle entanglement. These data are supplemented with recorded stranding locations of marine turtles and observed ghost net abundance. This study considered both likelihood and consequence of entanglement at the scale of the Northern Territory Government jurisdiction. Another study by Schuyler et al., (2016) presents the global risk of plastic ingestion to sea turtles, using one value for each “Regional Management Unit” (RMU; Wallace et al., (2010)). An RMU is a conservation unit of a species of sea turtle, which takes into account, genetics, mark-recapture studies, and nesting beaches, as a method of comparing threats among and between species of sea turtles. Assessing the risk of ingestion to turtles across a whole RMU is one of the intentions of the management unit framework, however, plastic distribution is so patchy and variable (Law et al., 2014) that it may be inappropriate in this case – because management of inputs of plastic, or removal must happen at smaller geographic or jurisdictional scales. While RMU-scale data is useful for understanding the global state of risk for turtles, RMU-scale data do not provide adequate spatial resolution for local management because the abundance of turtles and threats will not be uniform across the spatial extent of the RMU. Schuyler et al., (2016) present consequence of ingestion data from a literature review but they did not align the data with RMU, and did not consider it in the risk assessment, hence the true risk is not mapped at RMU scale. Both of the above examples look at only one aspect of the plastic/turtle interaction. Foraging turtles are exposed to the potential of both entanglement and ingestion interactions with plastics, and both must be taken into account to fully understand the risk plastics pose to sea turtles. Even where the plastic/animal interaction takes place at a local jurisdictional scale, it is still poorly understood (Vegter et al., 2014; Nelms et al., 2015).

Managers need information in a reasonable timeframe to take timely, proactive management action. The time necessary to complete field sampling and sample processing over a jurisdictional area to capture the variability would mean the outcome would not be readily available for decision makers at the time the data are needed. A method that will provide robust data on distributions of plastics at an appropriate spatial and temporal scale, in a timeframe appropriate for management action, is therefore necessary for practical management application.

1.2 Dispersal of pollutants in the marine environment

As explained above, accurate quantification of the distribution of pollutants is a key component of the likelihood of an interaction that causes harm. In a spatial approach this would involve understanding where the threat is located in time and space. Gaining observation data of distributions is difficult especially in the marine environment because our oceans are large, homogenous and relatively inaccessible (Ban, 2009; Browne et al., 2011) in comparison to terrestrial systems. Consequently, marine field data are more expensive and time consuming to collect relative to the amount of data obtained.

There are many methods for tracking objects or water masses at sea, for example satellite imagery or hydrodynamic modelling. Satellite imagery is useful for remote applications and for collecting data over large geographic areas, and is often used in water quality monitoring and the study of ocean productivity (e.g. Harvey et al., 2015). However, if the object/substance does not have an irradiance signature, or is not in high enough concentration to produce a detectable irradiance signature, it has limited utility. To fill this gap, hydrodynamic modelling is an alternative approach to understand pollutant distributions and is inexpensive compared with field sampling but can still be used in remote environments (e.g. Andutta et al., 2012). There has been a rapid expansion in the use of modelling in scientific and industrial fields due to the ready accessibility of high-performance computing and the improved performance of personal computing, along with the accessibility of physical input data such as winds and tides (Peng, et al., 2013).

Modelling of water movement is a useful tool in the marine environment as it allows scientists and managers to get an understanding of what is happening without the very costly (in time and resources) activities of extensive field surveys. Modelling is used to address many questions about dispersal in the ocean, for example, oil spill predictions (Guo and Wang, 2009; Cucco and Daniel, 2016; Spaulding, 2017), finding items lost at sea (Griffin et al., 2016), and floodplume modelling (Delandmeter et al., 2015). Modelling is also used for many ecological questions, for example turtle hatchling (Hamann et al., 2011; Wildermann et al., 2017) and larval dispersal (Andutta et al., 2012), and, of particular interest to my thesis, it is increasingly being used to understand the dispersal of plastic pollution (Yoon et al., 2010; Lebreton et al., 2012; Zhang, 2017).

Hydrodynamic models are inherently spatially explicit, and can be used to assess dispersal accumulation areas (or "hotspots") of plastic pollution. However, most of the existing models used for plastic pollution examine patterns at large geographic scales, for example oceanic basins (Lebreton et al., 2012; Maximenko and Hafner, 2012; Reisser et al., 2013; Ebbesmeyer et al., 2007; also see review by Kubota et al., 2005), or at coarse resolution within regional seas (Kako et al., 2011 Pichel et al., 2012). To accurately predict areas of accumulation at a management-relevant scale, fine-scale spatial resolution is required (100s of meters to kilometres). However, the scales and resolution of existing models range from: whole ocean modelling with a coarse resolution of 0.5 degree (~56 km at the equator) (Yoon et al., 2010; Maximenko et al., 2012) to a single basin with a finer resolution of $^{1}/_{12}$ degree (~9 km at the equator), for example, the East China Sea as in Isobe et al., (2009) and the Coral Sea as in Maes and Blake (2015). The smallest scale of a single coastline, with variable resolution, was the Queensland Coast (Australia) in Critchell et al., (2015) and the Gulf of Mexico in Nixon and Barnea (2010).

It is now clear that, while studies at large scales are useful, modelling of plastics in the coastal zone at small scales needs to take into account not only the physical processes (wind and tide), but also the biochemical processes (e.g., biofouling and degradation) specific to plastics (Zhang, 2017). These processes are not included in many of the models described above, and the inclusion of plastic-specific processes into the modelling could vastly improve our ability to understand movements and accumulation. Many of the processes are not relevant at larger oceanic scales (e.g. island wind shadow), however, to model dispersal and accumulation at small scales, they must be taken into account. At smaller scales it becomes important to know the fine-scale water movements driving the movement of plastics and thus fine-scale knowledge of processes are important. To model physical processes at this scale, a very fine resolution model is necessary. One such model is The Second-generation Louvain-la-Neuve Ice-ocean Model (SLIM; www.climate.be/slim).

The SLIM is a depth-averaged, two-dimensional, finite element model with variable resolution developed by Lambrechts et al., (2008). It has been used for a variety of physical and ecological modelling tasks including: fine sediment, fish larvae, floating debris, and turtle hatchling dispersal (e.g. Lambrechts et al., 2008 and 2010; Hamann et al., 2011; Andutta et al., 2013;

Critchell et al., 2015). The variable resolution (down to 100 m resolution) makes the model particularly useful in shallow coastal zones with complex bathymetry and topography. This model allows for fine-scale horizontal resolution and reduces the computational effort necessary to represent the whole model domain (Figure 1.1). Importantly, the flexibility of the SLIM means that the model domain can vary from the scale of the whole GBR, to one bay, enabling highly specific analysis to be completed in areas of complex topography such as the Whitsunday region of the GBR (see Box 1.1).

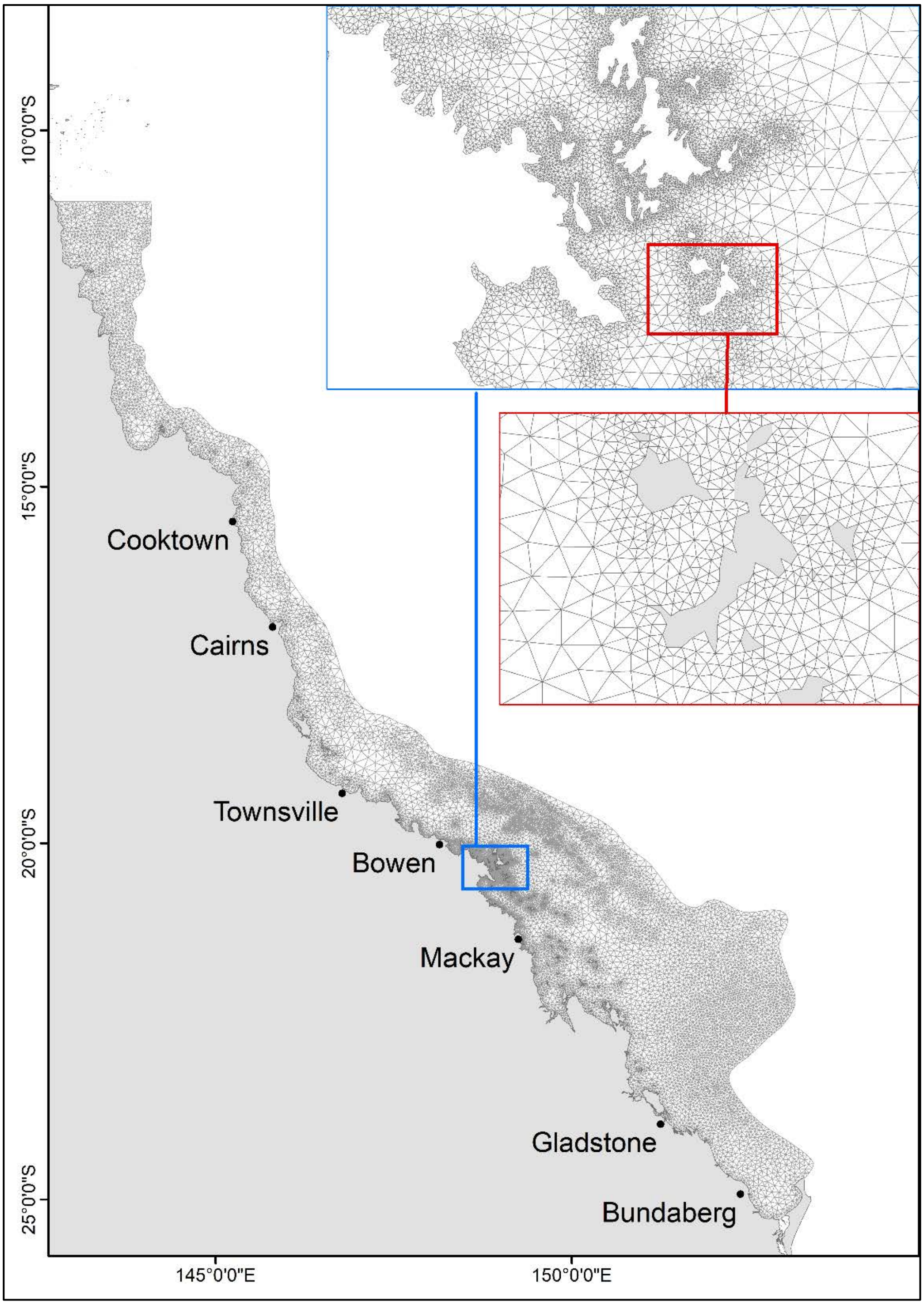

Figure 1.1: *Map showing the SLIM mesh across the model domain and at two scales of zoom (insets).*

Box 1.1: Study Site Description: The Whitsundays

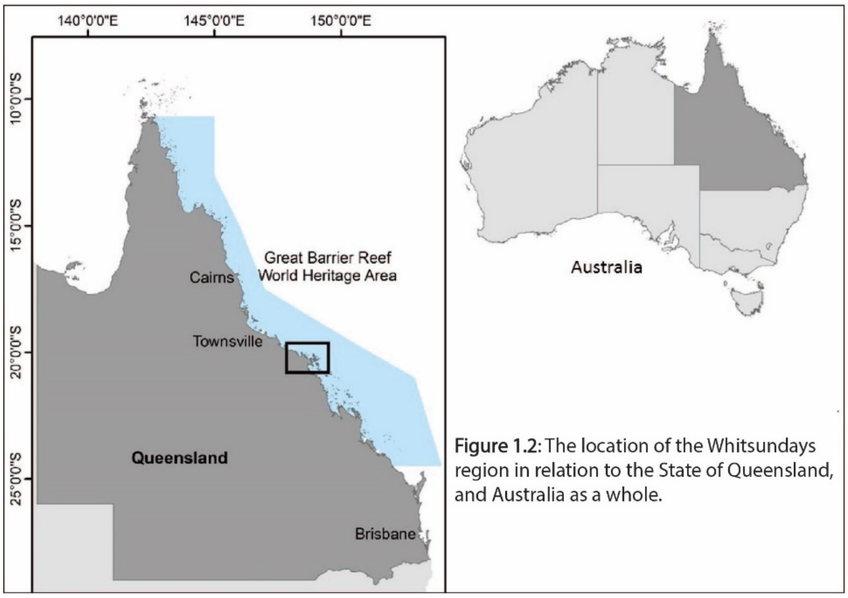

Figure 1.2: The location of the Whitsundays region in relation to the State of Queensland, and Australia as a whole.

The Whitsunday region of the Great Barrier Reef (20° S, 149° E) is in central Queensland, in the dry tropics region of the East Coast of Australia. Although the resident population is around 13,000 (Census, 2016), the region is one of the key tourism areas of the Great Barrier Reef and it receives around 500,000 tourists a year (National Visitor Statistics 2016). The Whitsunday area is important, economically and environmentally, therefore understanding the risk of plastics to this region will help managers prepare for future plastic abundance scenarios, e.g. calculating losses to tourism, and the necessity of intervention strategies.

The land associated with the Whitsunday region includes three main regional centres, Mackay, Proserpine and Airlie Beach. These regional centres are across two local council municipalities. The councils are responsible for waste management, and each council has independent waste management infrastructure making a combined management strategy for plastic pollution reduction in the Whitsunday region particularly challenging.

The region has two main seasons: Wet season (Monsoon Season, October to March), and the dry season (Trade Wind Season, April to September). The vast majority of the rain received in the region falls in the monsoon season, falling on three mainland river catchments; the Don (3690 km^2), the Proserpine (2530 km^2), and the O'Connell (2390 km^2).

The coastline consists of rocky shores, fringing reefs, mangrove, and lagoon environments. The subtidal areas consist of reefs, sandy bottom, and offshore trenches. The tidal range of three to five metres (Short and Woodroffe, 2009), creates strong tidal currents in the passages between islands and reefs and there are many small-scale jets and eddies in the region created by the complex hydrography. The complex and varied environment provides many potentially vulnerable habitats to use as case studies in my risk assessment approach.

1.5 Thesis objectives

The goal of this thesis is to understand the dispersal and risks of plastic pollution at a local management relevant scale. Therefore, in this thesis, I use a multidisciplinary approach to gain new insight into all aspects involved in the risk of plastics. Specifically, I aim to: 1) understand the dominant processes involved in plastic movement and accumulation in the coastal zone; and 2) explore the risk of plastic exposure to vulnerable habitats and species in the tropical coastal zone. This thesis contains six chapters, with four data chapters, each of which address a specific objective towards the two aims described above. Chapters 2 and 5 have been published. The data chapters were written to serve as stand-alone papers but I have standardised the formatting to make a coherent body of work.

Chapter 1 - Provides background information and identifies specific knowledge gaps that the thesis aims to resolve.

Chapter 2 - Aims to understand the dominant physical processes involved in plastic movement and accumulation in the coastal zone. As noted above, existing hydrodynamic models of plastic dispersal exclude pertinent properties of plastics (e.g. degradation, resuspension from the coastline, and wind-drift), which may affect their dispersal and accumulation. In this chapter I introduce a plastic-specific hydrodynamic model and conduct a sensitivity analysis to explore the new processes included in the model. The sensitivity analysis I undertook aims to understand which of the processes have the most influence on plastic dispersal and accumulation.

Citation: Critchell, K. & Lambrechts, J., 2016. Modelling Accumulation of Marine Plastics in the Coastal Zone; What Are the Dominant Physical Processes? Estuarine, Coastal and Shelf Science *171***,** *111-122.*

Chapter 3 - Aims to understand the risk of plastics to various vulnerable habitats and species in a spatially explicit approach to exposure. As noted above, the risks of plastics in the coastal environment are little understood, especially to determine where in the environment the risk is concentrated. In this chapter I use the new plastics model from Chapter 2 to create concentration distributions for plastics in the Whitsunday region and use these to inform spatial risk assessment for both habitats and species in the region, specifically coral reef, mangrove habitat, and flatback turtles.

Citation: Critchell, K. Hamann, M., Wildermann, N., & Grech, A. in prep. A spatially explicit exposure analysis of plastic pollution

Chapter 4 - Aims to use field data to interrogate the model in an attempt to further understand which hydrodynamic processes have the most influence on plastic dispersal and accumulation, at the small scale relevant to management, and to explore the sources of plastics in the Whitsunday region. This chapter builds on the knowledge gained in Chapter 2, which identifies the source of the plastics as the dominant key driver of accumulation hotspots. In this chapter I use field data for macro- and microplastics to compare with the model outputs of various modelled scenarios to understand the processes behind the observed accumulation.

Citation: Critchell, K., Hoogenboom, M., Grech, A., & Wolanski, E. & Hamann, M. in prep. Using field data to interrogate a plastics dispersal model.

Chapter 5 - Aims to understand the consequence of plastics exposure to marine life, using a planktivorous reef fish as a study species. I present the results of an experiment of microplastic exposure to the planktivorous reef fish *Acanthochromis polyacanthus*. The experiment: 1) explored the effect on body condition in the scenario of plastics replacing a natural food source; 2) assessed the scenario of microplastics in addition to food source; and 3) tested the amount of consumption at three plastic particle sizes for two size classes of fish.

Citation: Critchell, K. & Hoogenboom, M. under review. Effects of microplastic exposure on the body condition and behaviour of planktivorous reef fish (Acanthochromis polyacanthus)). PLOS One

Chapter 6 - Provides a synthesis of the thesis results, and discusses the limitations and implications of the work.

All citations used in the thesis are included in a collective reference list at the end of the thesis, beginning on page 139.

Chapter 2
Modelling accumulation of marine plastics in the coastal zone; what are the dominant physical processes?

Anthropogenic marine debris, mainly of plastic origin, is accumulating in estuarine and coastal environments around the world causing damage to fauna, flora and habitats (Chapter 1). Plastics also have the potential to accumulate in the food web, as well as causing economic losses to tourism and sea-going industries. If we are to manage this increasing threat, we must first understand where debris is accumulating and why these locations are different to others that do not accumulate large amounts of marine debris. This chapter demonstrates an advection-diffusion model that includes beaching, settling, resuspension/re-floating, degradation and topographic effects on the wind in nearshore waters to quantify the relative importance of these physical processes governing plastic debris accumulation. The aim of this chapter is to prioritise research that will improve modelling outputs in the future. I have found that the physical characteristic of the source location has by far the largest effect on the fate of the debris. The diffusivity, used to parameterise the sub-grid scale movements, and the relationship between debris resuspension/re-floating from beaches and the wind shadow created by high islands also has a dramatic impact on the modelling results. The rate of degradation of macroplastics into microplastics also has a large influence on the results of the modelling. The other processes presented (settling, wind-drift velocity) also help determine the fate of debris, but to a lesser degree. These findings may help prioritise research on physical processes that affect plastic accumulation, leading to more accurate modelling, and subsequently management in the future.

Citation: Critchell, K. & Lambrechts, J., 2016. Modelling Accumulation of Marine Plastics in the Coastal Zone; What Are the Dominant Physical Processes? Estuarine, Coastal and Shelf Science 171, *111-122.*

2.1 Introduction

The input and accumulation of anthropogenic marine debris such as plastics, is regarded in the public domain as an environmental and economic hazard. Macroplastic pollution (items larger than 5mm) accumulating on the coastline can affect tourism revenue (Jang et al., 2014) and the coastal habitat (Carson et al., 2011). The consumption of plastics, can cause damage to individual animals (Laist, 1997; Gregory, 2009; González Carman et al., 2014; Setälä et al., 2014) and have effects on the food chain (Boerger et al., 2010; Farrell and Nelson, 2013). There is evidence that microplastics (<5mm diameter) consumed by low trophic level species are transferred up the food chain as they are consumed by other trophic levels (Farrell and Nelson, 2013; Setälä et al., 2014). For these reasons it is important to create management action to prevent plastic waste from entering the environment, and there is a need to devise efficient debris removal schemes. While considering the importance of these factors, there are few data about the way different types of debris move in the ocean, why it accumulates in some locations more than others, and which parameters influence this most.

To maximise effectiveness of plastics debris removal for management and government agencies, geographic prioritisation of removal efforts must be considered. Oceanographic modelling is appropriate as part of a larger strategy to implement prioritisation and management (McElwee et al., 2012). The resolution required to accurately predict areas of accumulation at a beach scale is quite fine, ranging from a few 100 meters to 1 km. However, the recent models of plastic movement in the marine environment focus on models examining much larger scales, for example oceanic scales (Lebreton et al., 2012; Maximenko and Hafner, 2012; Reisser et al., 2013; Ebbesmeyer et al., 2007; also see review by Kubota et al., 2005), or within seas (Kako et al., 2011 Pichel et al., 2012) at coarse resolution. The scales and resolution of plastic movement models range from whole-ocean modelling with a coarse resolution of 0.5 degree (Yoon et al., 2010; Maximenko et al., 2012) to a single basin with a finer resolution of $^{1}/_{12}$ degree i.e. the East China Sea as in Isobe et al., (2009) and the Coral Sea as in Maes and Blake (2015). The smallest scale of a single coastline, with variable resolution was the Queensland Coast (Australia) in Critchell et al., (2015), and the Gulf of Mexico in Nixon and Barnea (2010). One reason for the large spatial scales is the time over which the simulations are run. The time scales varied from 30 years of simulations as in Lebreton et al., (2012) to a few weeks as in Carson et al., (2013) and Critchell et al., (2015).

Modelling plastics in the ocean can be challenging since plastics range in size, shape, buoyancy, density, etc. To avoid this issue, some studies model a specific type of plastic: Ebbesmeyer et al., (2007) modelled a cargo spill (tub toys); Kako et al., (2011) modelled bottle caps; Ebbesmeyer et al., (2011) modelled crab pots; and Isobe et al., (2014) studied different sizes of plastic and how they move in on-shore and off-shore directions. Many studies, however, continue to model plastics as a general category (Isobe et al., 2009; Martinez et al., 2009; Yoon et al., 2010; Hardesty and Wilcox, 2011; Kako et al., 2011; Lebreton et al., 2012; Maximenko and Hafner, 2012; Maximenko et al., 2012; Reisser et al., 2013; Maes and Blanke, 2015; Critchell et al., 2015).

Specialist, large-event debris models have also been developed. For example, the National Oceanic and Atmospheric Administration (NOAA) marine debris probability model developed for hurricane debris in the Gulf of Mexico, uses 100 m grid cells to compute probability of debris being found after a hurricane. Parameters such as wind speed, storm surge and infrastructure were used to assess the probability (Nixon and Barnea, 2010). A model for the debris from the 2011 Japanese Tsunami has also been developed by Maximenko et al., (2015), who used four different modelling systems with resolution from $^1/_4$ to $^1/_{12}$ of degree grid. The methodology used for oil spills has been found to be effective for modelling floating plastic debris (Le Hénaff et al., 2012), where the floating plastic is assumed to have a velocity equal to the vectoral sum of the water currents and the wind-drift velocities. The direct movement of plastics due to the wind (wind-drift) is neglected in many studies that model the movements of plastic in the ocean (Isobe et al., 2009; Martinez et al., 2009; Kako et al., 2011; Reisser et al., 2013; Isobe et al., 2014; Maes and Blanke, 2015). In studies that include wind-drift, the value of the wind-drift coefficient varies from 1% (Ebbesmeyer et al., 2011) to 6% (Maximenko et al., 2015), and in some studies a range of values are used or the value used is not given but instead the empirical formula for calculating the wind-drift is given (Kako et al., 2010). Submerged plastic debris is spread through the water column, with no exposure to the wind and hence no wind-drift is assumed (Reisser et al., 2013).

For a model to become realistic and useful, it not only needs to apply the oil-spill model methodology, like that of the GNOME (Beegle-Krause, 2001), OSCAR (Reed et al., 1995), and other model types reviewed by Potemra (2012), but also needs to include a number of additional processes specific for plastics, which so far appear to have been neglected in marine debris models. These processes are sketched in Figure 2.1, and include: (1) degradation of macroplastics into secondary microplastics; (2) the different wind-drift coefficient for macroplastics (that tend to float) and microplastics (that experience no wind-drift as they tend to be in suspension in the water column; Reisser et al., 2013); (3) rates of settling; (4) burial in beaches; (5) resuspension or re-floating from beaches; and (6) the non-uniformity of the wind near the coast, especially the dramatically reduced wind velocities behind hills on the land (wind shadow). The incorporation of these parameters into a model should improve the ability to predict the movement and the fate of plastics at coastal scales. This improved and more robust model could be used for plastics in a similar manner to oil-spill models for oil slicks. The oil-spill model methodology is basically advection-diffusion models coupled with chemical sub-models of the weathering of the oil, and are now routinely used by industry management (Chao et al., 2001; Tkalich et al., 2003; Guo and Wang, 2009). Such a model methodology is needed to improve predictions of debris accumulation and thereby improve management strategies for debris removal and mitigation. In addition, the improved model may also be used to backtrack and ultimately help to locate the sources of plastic pollution arriving at a given location, which would also support management goals (Reisser et al., 2013; Thiel et al., 2013). In order to work towards this, true values for the parameters described above must be experimentally determined or found through field observations.

In this study, I develop and explain a plastic oceanographic model to study the fate of plastics in estuarine and coastal waters (within 100 km of the coast). I demonstrate the application of this model in the complex case of a rugged coastal region with shallow waters and numerous islands and headlands. The basis of this plastics oceanographic model is a high resolution oceanographic, advection-diffusion model that also includes all the processes identified in Figure 2. 1. I propose a simple method to assess and rank the relative influence of these various physical processes on the movement of plastics in the coastal zone, using this method to prioritise research of the physical processes influencing plastic movements at sea.

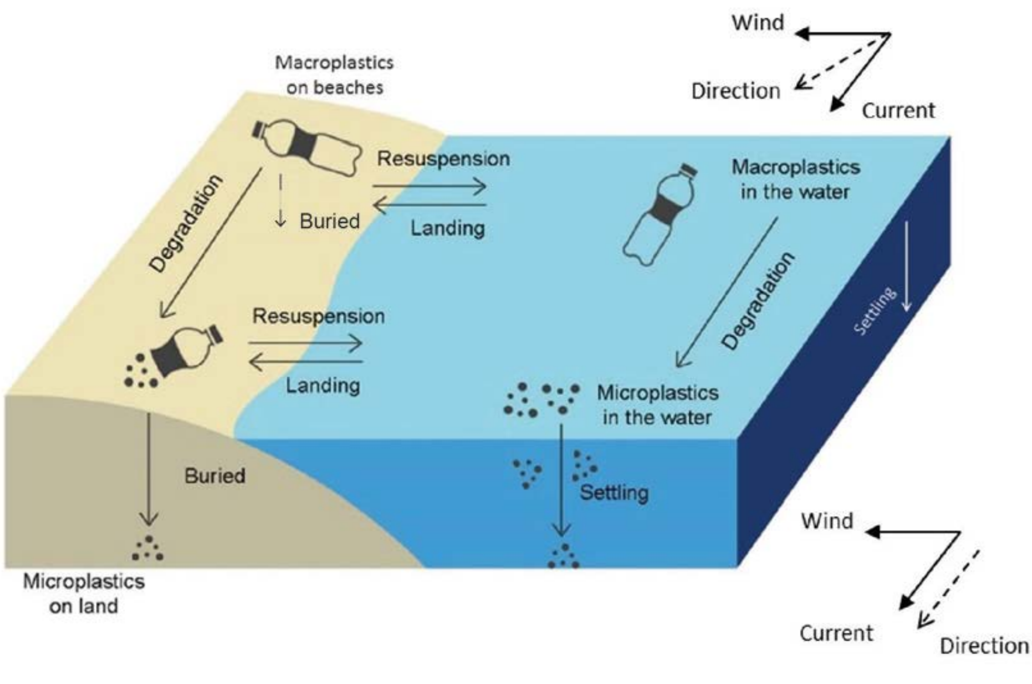

Figure 2.1: *Schematic of the physical process pathways plastic items undergo when dumped at sea.*

2.2 Methods

2.2.1 The oceanographic model

To evaluate the relative importance of coastal processes on the movement of marine debris, I conducted a sensitivity analysis using the Second-generation Louvain-la-Neuve Ice-ocean Model (SLIM; www.climate.be/slim). SLIM is a depth-averaged, two-dimensional, finite element model with variable resolution developed by Lambrechts et al., (2008), which has been used for a variety of physical and ecological modelling tasks including: fine sediment, fish larvae, floating debris, and turtle hatchling dispersal (e.g. Lambrechts et al., 2008 and 2010; Hamann et al., 2011; Andutta et al., 2013; Critchell et al., 2015). The variable resolution (down to 100 m resolution) makes the model particularly useful in shallow coastal zones with complex bathymetry and topography. This model allows for fine-scale horizontal resolution and reduces the computational effort necessary to represent the whole model domain. The appropriate use of a depth-average model in shallow, vertically well-mixed waters was previously explored by Critchell et al., (2015). In that study, it was shown that, in well-mixed shallow water environments, the diffusion patterns of particles are very similar at the surface, middle and bottom of the water column, and the use of a three-dimensional modelling approach, (which is computationally very expensive) may be an unnecessary use of computational effort.

The study region used to conduct the sensitivity analyses was the Whitsunday region of the Queensland coast, and is part of the Great Barrier Reef Marine Park (20.2^{0}S, 149.0^{0}E; Figure 2.2). This region is made up of approximately 74 coastal islands, coral reefs and other marine and coastal habitats. The coastal waters are primarily shallow with a mean depth <20 m (Figure 2.2). This region is also a tourism centre, making it economically important not only for Queensland but also for Australia as a whole. The area has had a marine debris removal program run by Eco-Barge Clean Seas Inc. since 2009. The islands and reefs create high levels of topographic and hydrodynamic complexity and create a large variety of unique locations with a rugged coastline, providing an ideal situation to quantify the relative importance of the various processes controlling the fate of plastics in complex topography and bathymetry.

In the model, all simulated plastics are seeded as macroplastics on the water surface, directly affected by the wind. From this state, an individual plastic can go through a series of pathways: beach, settle to the sea floor, degrade into microplastic, or continue as a wind driven macroplastic on the surface of the water. A macroplastic that has beached can then be re-floated, degrade into microplastic or remain beached. A microplastic in suspension can either beach, settle or continue in suspension, while not being directly be affected by the wind. A microplastic on the beach is assumed to be able to be resuspended. Simulation is discontinued for macro- or microplastics that have settled onto the sea floor and are assumed not to be resuspended (Figure 2.1). Simulation is also discontinued for macro- and microplastics that leave the model area through the open sea boundaries and are assumed not to be transported back into the study area. When the wind pushes either macro- or microplastic particles on a coastline, those particles are considered as beached. Beached particles do not move but have a chance to be resuspended/re-floated at each time step (5 minutes). Resuspended particles are placed at a random position in the neighbouring cell. In some scenarios, the resuspension rate is constant, in others beached particles in sheltered area (i.e. downwind coastlines) are assumed not to be resuspended.

Primary microplastics (plastics that are made and designed to be very small) are not included in this sensitivity study, as this added a complicating factor, but they should be considered in any predictive modelling undertaken in the future. At each time-step, macroplastics are assumed to be able to degrade to secondary microplastic at a given degradation rate. The degradation rate is a constant fixed for each scenario. Unlike macroplastics, microplastics are assumed to be spread homogenously throughout the water column. Microplastics are assumed not to be directly affected by the wind and the dispersion process incorporates the influence of the gradient of the bathymetry (see Heemink, 1990; Spagnol et al., 2002; Deleersnijder, 2015). The model allows for the chance for all plastics to settle to the sea floor. When this happens, those particles are assumed not to be resuspended from the sea floor (Figure 2.1).

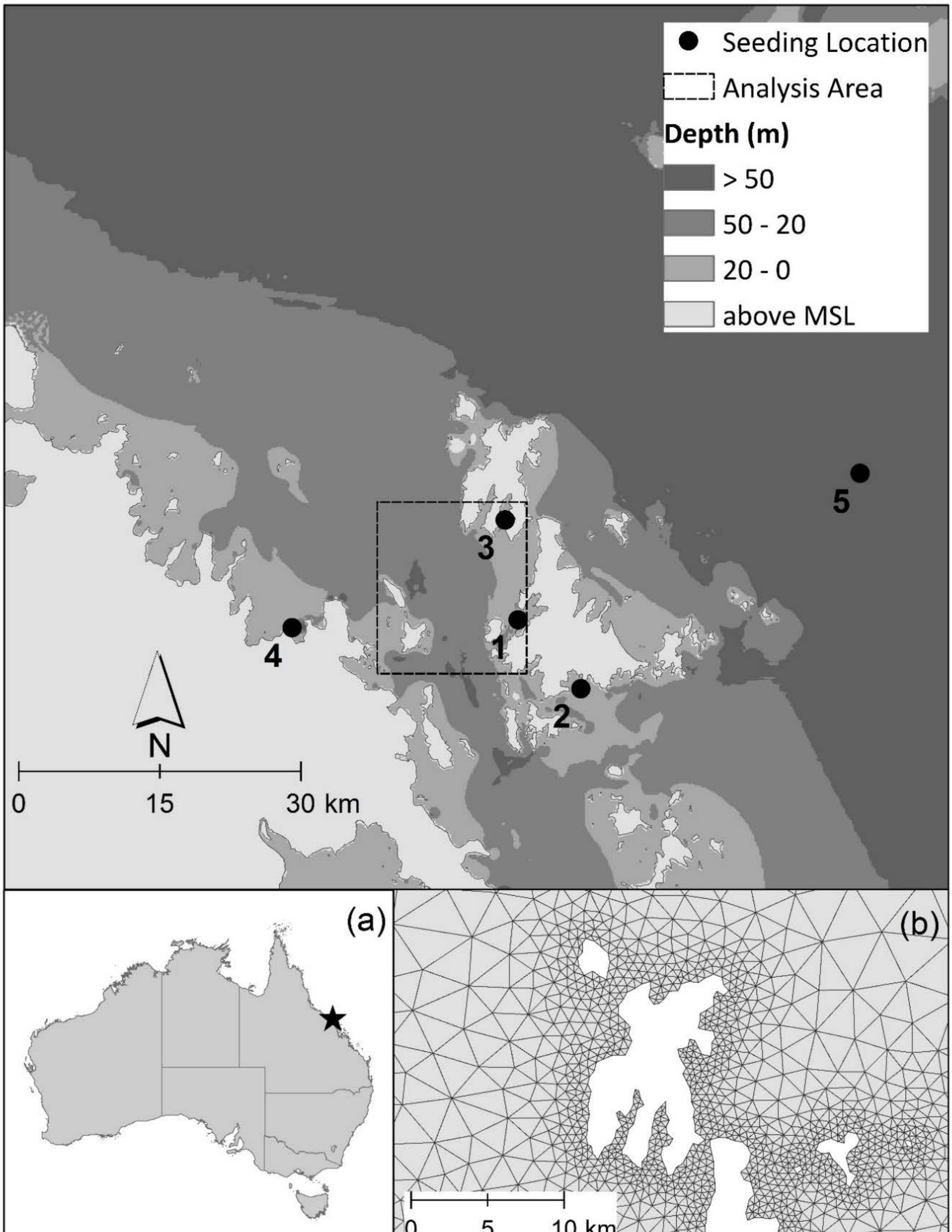

Figure 2.2: *Case study area of the Whitsunday region of the Great Barrier Reef shown by the star in the inset map (a). Inset (b) shows an image of the simulation mesh around Hook Island in the Whitsunday group. The main map indicates the analysis area used to calculate the comparative indices and the seeding locations used for the simulations, Site 1 is the standard location.*

2.2.2 The physical processes of drifting, beached and buried plastic

To quantify the relative importance of the source locations and the processes shown in Figure 2.1, a number of sensitivity scenarios were run. The specific parameters used for the sensitivity analyses along with key references are listed in Table 2.1 and the source locations are shown in Figure 2.2. The ‘standard run’ was set so that 10,000 macroplastic particles were released at each seeding site, with a wind-drift coefficient of 2%, a settling rate of both “macro-” and “micro-” plastic debris onto the sea bed of 0.2 day^{-1}, resuspension from the coast for both macro- and microplastic debris set to a rate of 0.2 day^{-1}, with the resuspension of beached particles being affected by the wind shadow (i.e. the resuspension is reduced in areas that experience wind shadow behind hills) and the wind shadow length was 2500 m. The degradation rate for both beached and suspended plastic particles was set to 0.2 day^{-1} for all sensitivity simulations, this is likely far higher than would occur in the natural environment, however, by using this value I can ensure that the scenarios are comparable and that microplastics were within the area of interest. Scenarios of best-estimate degradation rates are presented but not considered in the sensitivity analysis of physical parameters, they are analysed separately. Mixing at horizontal scales smaller than the mesh size was parameterized by a turbulent diffusion coefficient (Okubo, 1971). The true value of this coefficient is unknown and probably site-specific. In a shallow, rugged bathymetry its value is believed to be in the range 5 - 20 $m^2 s^{-1}$ (Andutta et al., 2011; Hrycik et al., 2013): a value of 10 $m^2 s^{-1}$ was adopted as the standard value. As there is little published work determining the value of the majority of these parameters, the values used for the sensitivity analysis (Table 2.1) are best estimates of likely values. I chose these values so they are appropriate to use to compare between scenarios in order to make an assessment of the relative importance of the parameters in the model and in the fate of marine debris in the coastal zone. To test the sensitivity of the parameters, they were changed one by one, being halved and then doubled from their original values in the ‘standard run’, and the predicted fate of plastics was then compared to the fate of plastics for the ‘standard run’. The same parameter values of a ‘standard run’ were also used at the various source locations to test the effect of varying the source within a relatively small area.

***Table 2.1**: Parameters and values used in the sensitivity analyses*

Parameter	Values	Explanation and key references
Resuspension rate (day-^{1}) of buried plastic particles	0.1, 0.2, 0.4	Beached the particles may be resuspended on the next incoming tide (Johnson, 1989; Johnson and Eiler, 1999). Applied to both “macro” and “micro” particles independently. Unit is proportion per day. No data avaliable
Degradation rate (day^{-1}) from macroplastics to microplastics	0.2 (1x10^{-6} and 2x10^{-6} used in degradation specific scenarios)	Whole plastic items as well as fragments undergo physical and chemical degradation causing them to break up into many smaller pieces (decribed in Cooper and Corcoran, 2010; O'Brine and Thompson, 2010). Applied to both suspended and beached particles independently (Isobe et al., 2014). Unit is proportion per day. Limited data avaliable
Wind shadow (or lee effect) of hills affecting resuspension of particles	presence/absence	Lee effect is created when high islands block the wind, in the lee area waves are suppressed and resuspension is thus reduced (Myksvoll et al., 2012). No data avaliable
Length of wind shadow (m)	1000, 2500, 5000	The wind shadow affects the hydrodynamics because the extent to which wind pushes the water will be influenced by topography (Wolanski and Delesalle, 1995).
Wind-drift coefficient (only applied to macro-debris)	1%, 2%, 4%,	The degree to which the wind directly influences the individual particles (Daniel et al., 2002). This is only applied to the macroplastics, assumed to be floating on the surface. Limited data avaliable
Settling rate (day^{-1})	0.1, 0.2, 0.4	As items degrade or become water logged they will sink to the sea floor (Lee et al., 2006; Le Hénaff et al., 2012). This rate is applied to both “macro” and “micro” particles independently. Unit is proportion per day. No data avaliable
Source location	5 different locations (shown in Figure 2.2)	The sites are near the coast in open and closed waters upwind, and downwind. This is similar to testing the importance of the source with an oil spill model (Maximenko et al., 2012). Limited data avaliable
Diffusivity	5, 10, 20 m^2s^{-1}	The parameter in the model to account of mixing at scales smaller than the mesh size (Okubo, 1971, Hrycik et al., 2013). Limited data avaliable

To assess the effect of the seeding location on the results of the simulation, I used the standard parameter values from 4 more seeding locations. Site 1 (the standard location) and Site 4 were close to the coast on the lee side of the land, Sites 2 and 3 were on the exposed side of the land and close to the coast, and Site 5 was off shore, not affected by wind shadow (Figure 2.3).

2.2.3 Data analysis

For the scenarios examining the physical processes, the calculated latitude and longitude locations of the simulated plastics were extracted for each day of simulation using an analysis area around the source location (see Figure 2.2). The analysis area was set around the seeding location so that the edge of the analysis area was close to the coast, this was to reduce the area that would be impossible for simulated plastics to be in. The plastic particles (macro: beached, floating on the surface, settled, and micro: beached, suspended in the water column, settled) that remained in the analysis area (Figure 2.2) were used to create indices to compare the different scenarios through time. All the scenarios examining the physical processes were seeded from the standard location, within the analysis box. The scenarios assessing the effect of seeding location were compared using different indices, which did not use the analysis box, described later.

The following indices were calculated for each of the physical process scenarios: (1) residence time (days) of suspended macroplastics, to be able to compare the rate at which they leave the system; (2) residence time of beached macroplastics, to compare the time spent on the coastline in macroplastic state; (3) time for doubling (days) of microplastics on the beaches, using the number of microplastic particles on the beach by the end of day 1 as the starting value to evaluate the rate of increase of microplastic particles; (4) maximum number of suspended microplastics in the study area after initial release, to understand the change from the standard in settlement, resuspension, and drifting from the analysis area; (5) total accumulated macroplastics on the sea floor; and (6) total accumulated microplastics on the sea floor. These parameters were used to understand the rate of accumulation over the 8 day simulations.

For the scenarios testing the effect of source location, a set of indices were created to compare the scenarios to the standard run. The indices were: (1) the distance from the seeding location of the furthest and closest macroplastic particles; (2) the latitudinal spread of the microplastic particles; and (3) the concentration of particles on the coastline. The concentration of particles was calculated by counting the number of plastics per 100 m of coastline. The average concentration of the seven highest accumulating sections of coastline was used as the comparison index. This was to assess the degree of spread of accumulated plastics between scenarios.

The degradation process was analysed separately as it acts over a different temporal scale to the other processes. Four scenarios of degradation rates were assessed, where all parameters were as the standard run except for the degradation rate which was set to: (a) $1x10^{-6}$ day^{-1} for macroplastics at sea and $1x10^{-5}$ day^{-1} for macroplastics on the beach, run in the model for 1 month; and (b) $2x10^{-6}$ day^{-1} for macroplastics at sea and $2x10^{-5}$ day^{-1} for macroplastics on the beach, run for 1 month. Scenarios (c) and (d) are to assess the difference between degradation of macroplastics that are beached and at sea. In these cases, the degradation rate is set to $1x10^{-6}$ day^{-1} for one condition (beached or at sea) and set to zero for the other condition. To show the comparison of the temporal scales after 8 days (as in the standard run), the mean centre location of the microplastic particles was calculated in ArcGIS.

2.2.4 Ranking process

To understand the relative importance of each parameter in the model, a relatively simple method of comparison was adopted. Due to limited data and true parameter values, the modelling at this point cannot justify a more sophisticated or complex analysis. For each scenario, the absolute value of the 6 indices described above were made relative to their values in the standard run. The scenarios were ranked by the dimensionless difference to the standard. For example, if “scenario 4” has a negative difference from the standard, larger than all scenarios with positive differences, then “scenario 4” is ranked higher. The largest dimensionless difference was considered the most important scenario and the smallest difference was considered the least important. All the scenarios were ranked first by the six indices individually, then an overall rank for each scenario was taken as the median rank it received for the indices. The process rank was taken to be the median rank of the scenarios associated with the process (see Table 2. 1 for list of processes, Table 2. 2 for the scenarios). The median was chosen to lessen the effect of extreme ranks.

*Table 2.2: Summary table of the indices for each scenario relative to the standard run (** the level of beached microplastics in this scenario is decreasing due to the rate of resuspension). The scenario numbers refer to the labels of Figure 2. 4*

Scenario	Scenario Number	Residence time Macro-suspended	Residence time Macro-beached	Beached micro plastics time for doubling	Max number of micro suspended	Accumulated microplastics	Accumulated macroplastics
Standard	1	1	1	1	1	1	1
Macro- Settling 0.1	2	1.06	0.97	1.01	1.05	1.12	0.47
Macro- Settling 0.4	3	0.86	1.01	1.02	0.82	0.77	1.87
Macro- Resuspension 0.1	4	0.99	1.03	1.01	0.98	1.03	0.96
Macro- Resuspension 0.4	5	1.05	1.03	1.04	0.96	0.96	0.98
Micro Settling 0.1	6	1.01	1.02	1.01	1.07	0.57	0.96
Micro Settling 0.4	7	1.03	1.04	1.02	0.87	1.49	0.94
Micro Resuspension 0.1	8	0.99	0.99	1.02	0.94	0.94	0.95
Micro Resuspension 0.4	9	1.01	1.03	1.02	0.90	0.94	0.95
No Shadow Effecting resuspension	10	2.28	0.33	**-0.80	4.32	5.02	2.57
Wind Shadow 1000 m	11	1.11	0.99	1.00	0.86	0.79	0.88
Wind Shadow 5000 m	12	1.09	0.99	1.02	0.80	0.72	0.82
Wind Drift 1%	13	1.61	1.01	1.04	0.89	1.06	1.05
Wind Drift 4%	14	0.66	0.95	1.02	0.95	0.85	0.90
Diffusivity 5 m^2s^{-1}	15	1.25	1.18	0.96	1.52	1.59	1.34
Diffusivity 20 m^2s^{-1}	16	0.78	0.97	1.07	0.57	0.53	0.60

2.3 Results

Changing the seeding location from the standard to one of the alternate locations had by far the biggest influence on the model results. Seeding locations that face the dominant wind direction (Sites 2 and 3) had a smaller spatial distribution of suspended simulated plastics compared to the standard location. At Site 2 (Figure 2.3) all macroplastics were beached after 8 days of simulation and at Site 3 the macroplastics were distributed downwind 57km (mean distance). Sites in the lee of the islands (Figure 2.3; Site 1 and Site 4) had a larger spread of microplastics (almost 0.2 degrees larger) than the exposed sites (Table 2.3). The seeding location away from the coast (Figure 2.3; Site 5) had by far the largest latitudinal distribution of microplastics and the most variation with the distance the macroplastics moved (Table 2.3).

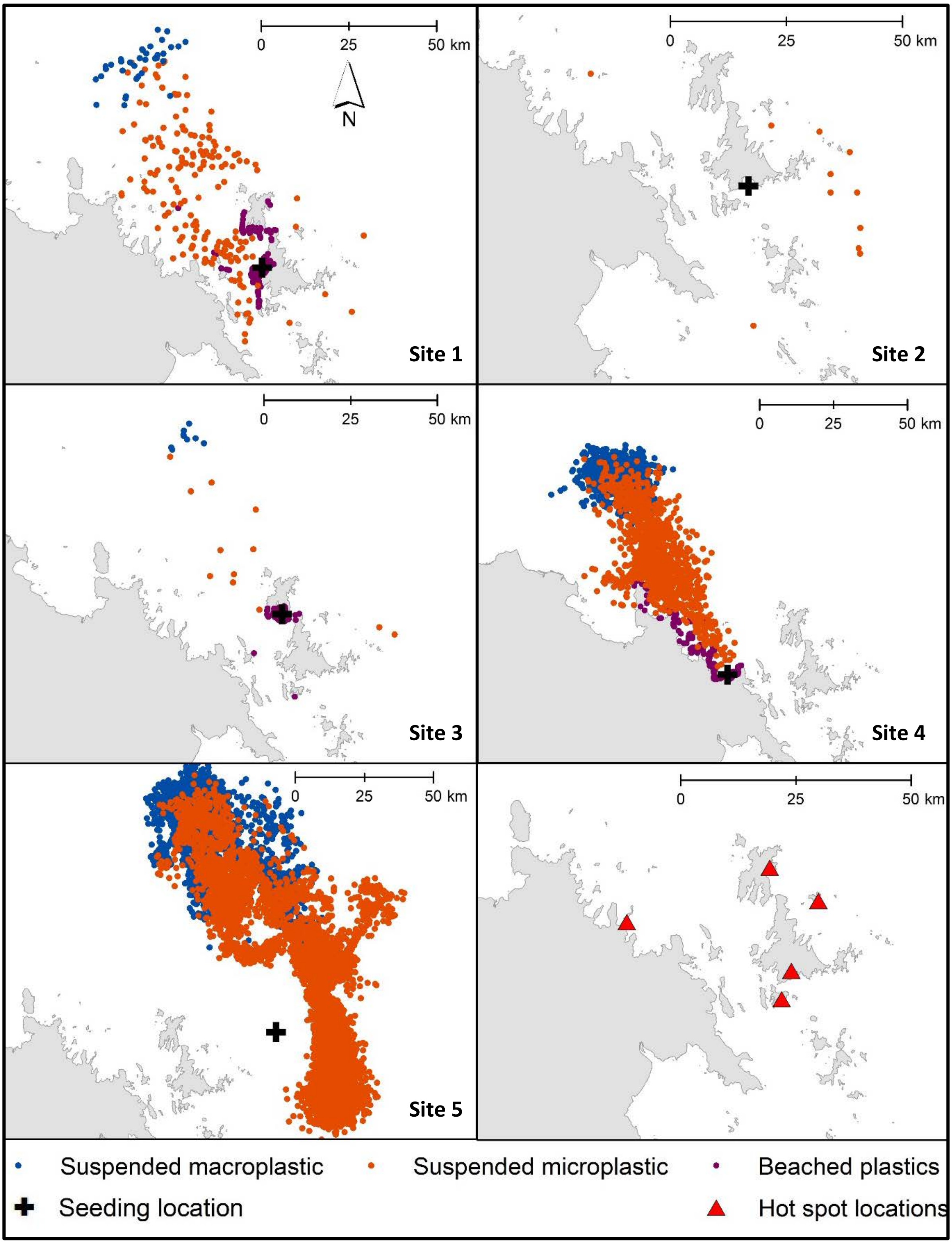

Figure 2.3: *Effect of seeding location on distribution of suspended and beached plastics after 8 days of simulation. The blue dots represent the macroplastics in suspension, the orange dots represent the microplastics in suspension and the purple represent the pooled macro- and microplastics that have landed on the beaches. The lower right panel shows the location of known hot spots for debris accumulation as described by Eco Barge Clean Seas Inc. (personal communications 2013).*

Table 2.3: *Summary table of the comparative indices for each location scenario*

location	Latitudinal spread of suspended microplastics (DD)	Distance range downwind from seeding location of suspended macroplastics (km)	Mean concentration of beached particles (count)
1 (standard location)	0.71	54.5 – 73.2	213
2	0.49	All beached	4417
3	0.46	53.7 – 60.4	2160
4	0.64	58.9 – 89.0	1309
5	1.18	33.0 – 100.6	None

From the standard location, both simulated macro- and microplastics accumulate in large numbers on the coastline close to the seeding location (Figure 2. 4; beached macroplastics and beached microplastics). Macroplastics are driven far from the seeding location in the same direction as the dominant wind, whereas microplastics in suspension spread against the wind direction as long as currents are favourable (Figure 2.4; suspended plastics). Plastics on the sea floor have a high concentration close to the seeding location (Figure 2.4; settled plastics).

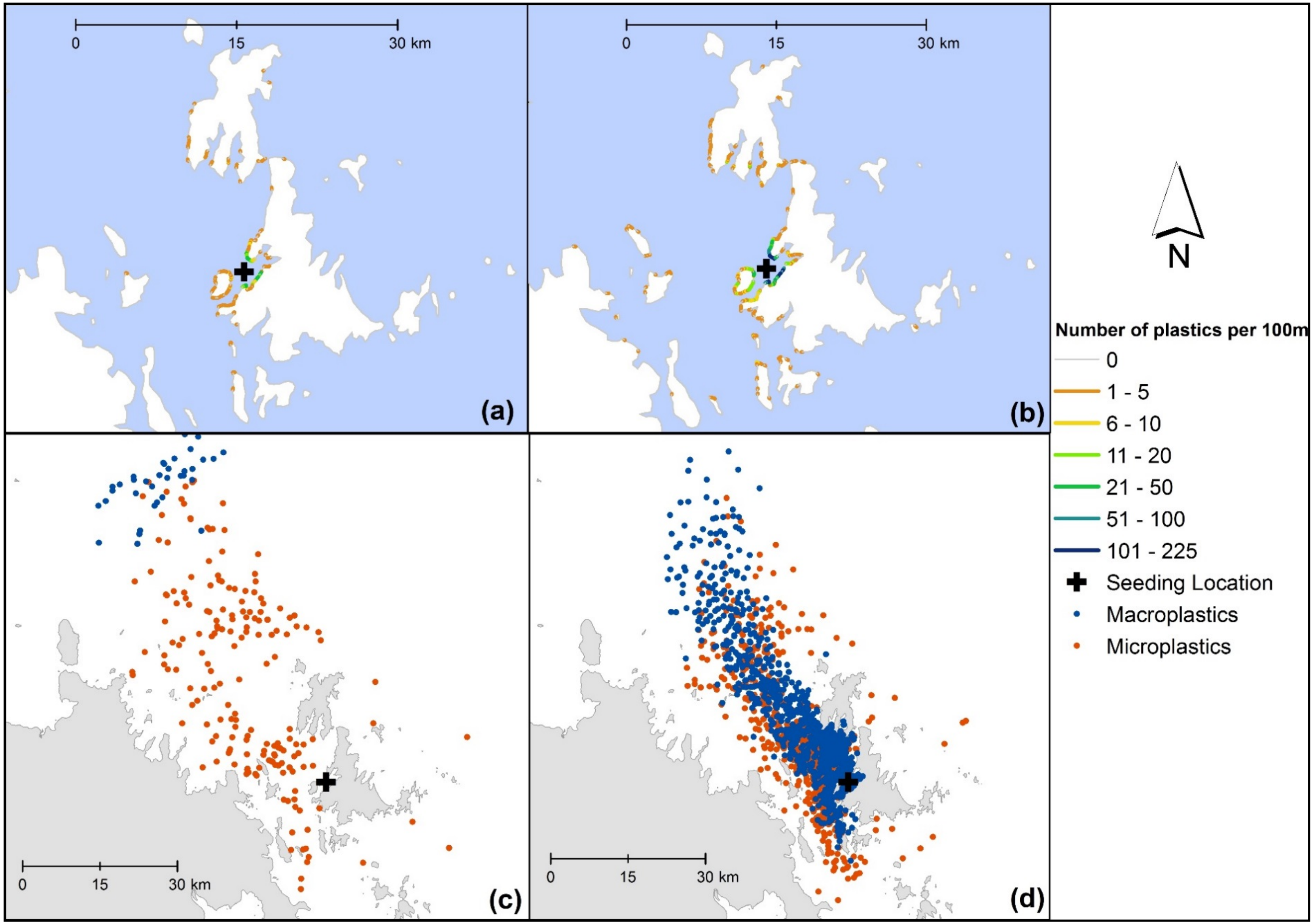

Figure 2.4: *Concentration of beached (a) macroplastics and (b) microplastics, and location of (c) suspended and (d) settled macro- and microplastics, after 8 days for the 'standard run'.*

The locations of simulated plastics in the degradation rate scenarios are shown in Figure 2.5. Under scenarios (a), (b) and (d) the vast majority of microplastics are distributed far away from the source location. After 8 days, the mean centre of the microplastic locations was 42.25 km downwind from seeding location compared with 26.3 km in the standard run. There would have been insignificant microplastics in the analysis area. In scenario (c), the degradation rate of macroplastics at sea was zero, therefore no microplastics were created.

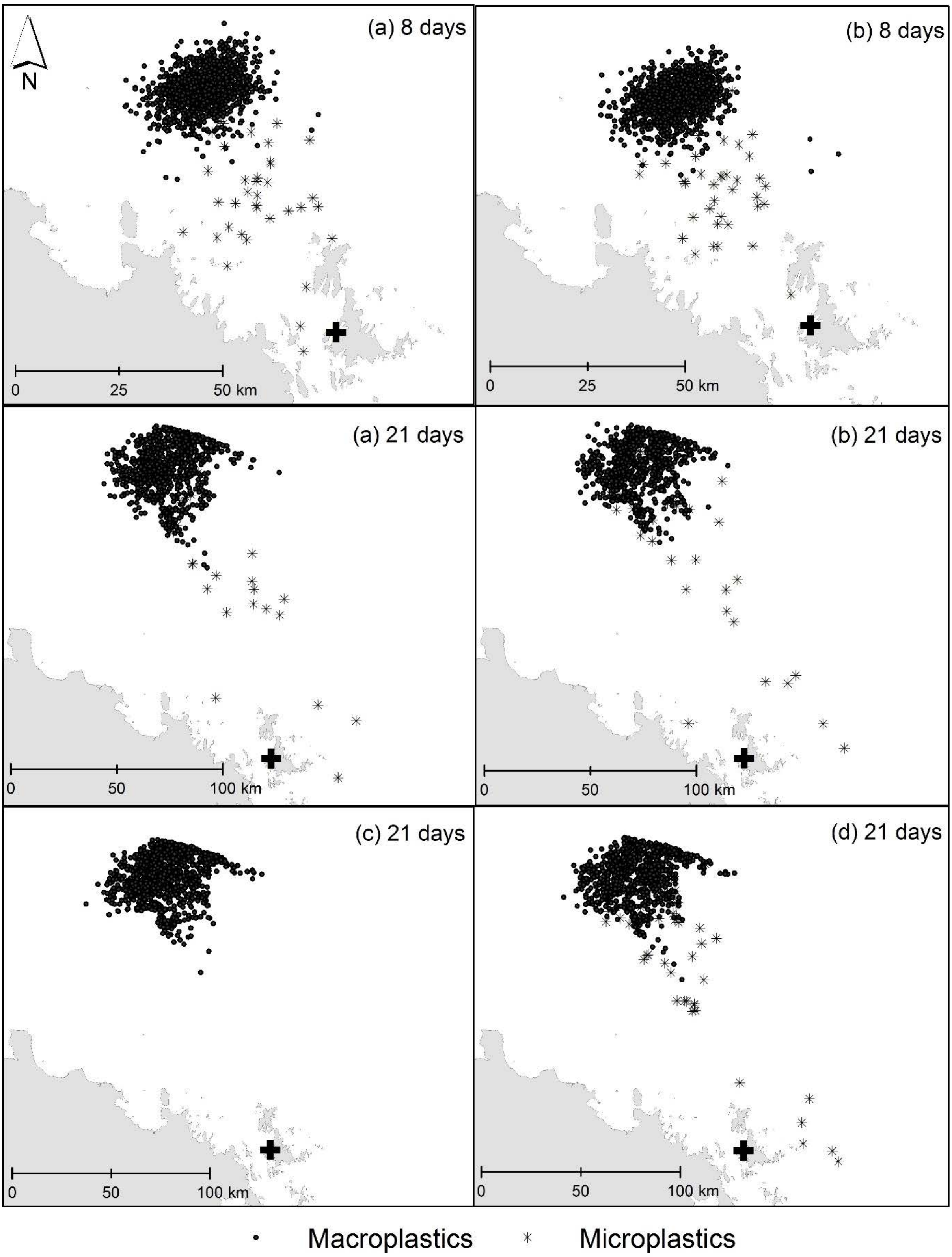

Figure 2.5: *The effect of degradation rate on particle locations after one month days at (a) 1x10-6 day-1 for plastics on the beach and 1x10-5 day-1 for plastics at sea. (b) 2x10-6 day-1 for plastics on the beach and 2x10-5 day-1 for plastics at sea. (c) 1x10-6 day-1 for beached particles, (d) 1x10-6 day-1 for plastics at sea. The star points represent microplastics, the dark circular points represent macroplastics and the seeding location is shown by the thick black cross.*

Scenario 10 (wind shadow does not affect resuspension of simulated plastics) is an outlier in this sensitivity analysis. In this scenario, the resuspension rate was set to 0.2 day^{-1} regardless of the presence of wind shadow, and caused a large difference in the result when compared to the standard run. There are suspended macroplastics in the analysis area throughout the 8 day simulation whereas the standard only had 2 macroplastics remaining in suspension at the end of day 4 (Figure 2.6). The maximum number of suspended microplastics in the analysis area in Scenario 10 is over 4 times that of the standard run. The number of simulated plastics beached on the coastline, either macro- or micro-, in Scenario 10 is orders of magnitude smaller than any other scenario (e.g. beached macroplastics in Scenario 10 after day 4 is 11 simulated items, the mean for the other scenarios is 1809 simulated items). The number of settled plastics inside the simulation area for Scenario 10 is over double that of the standard run (macro- 2.57x and micro 5.02x larger). When Scenario 10 is not considered, all other simulations have similar patterns. The number of suspended macroplastics in the analysis area approaches zero in all scenarios by the end of day 3, except when the wind-drift coefficient is reduced to 1% this scenario loses its last macroplastic particle on day 7.

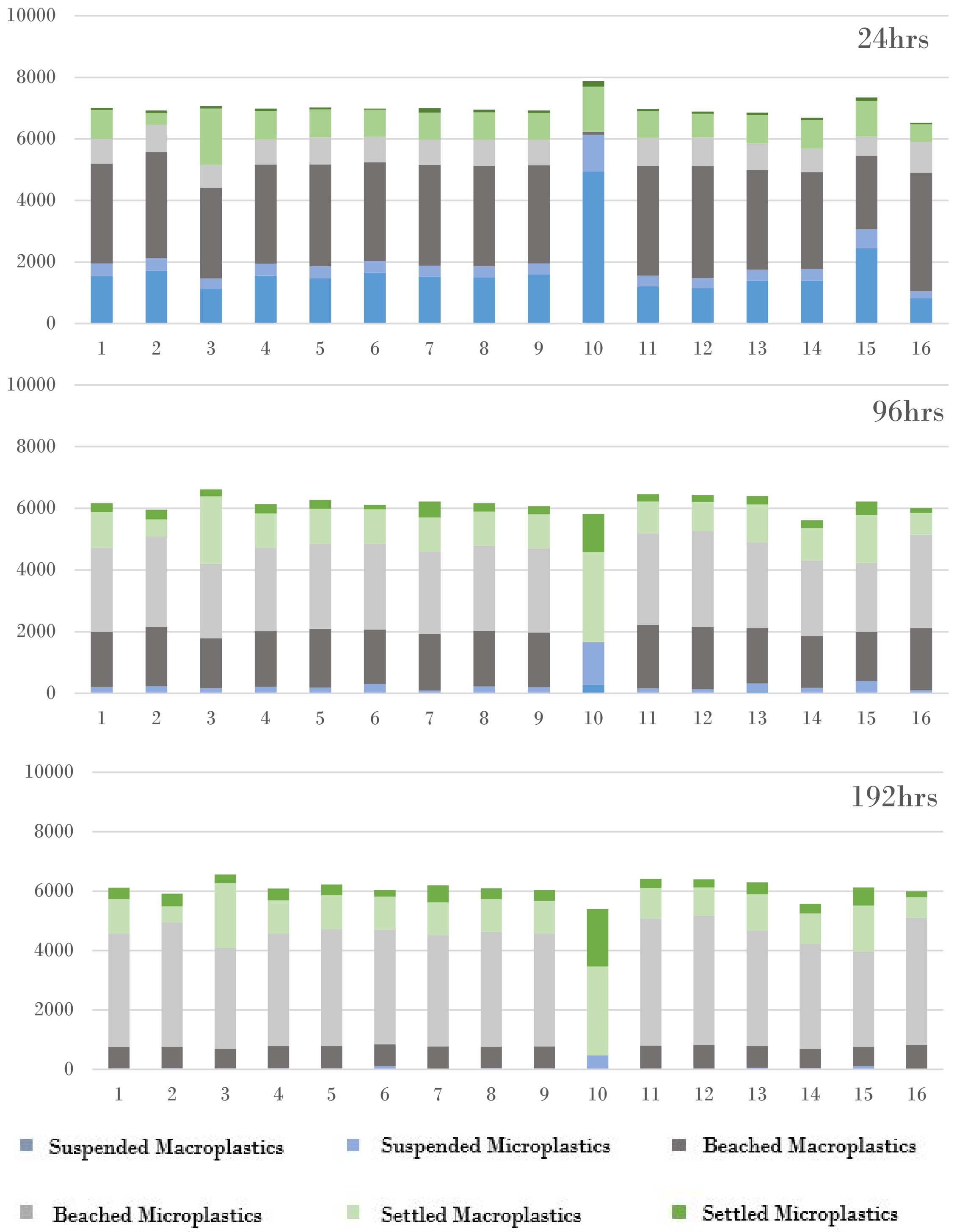

Figure 2.6: *The number of each particle type in the analysis area after days 1, 4 and 8 of simulation, for each scenario (the reader is referred to Table 2. 2 for scenario descriptions) each scenario was seeded with 10,000 macroplastics in suspension.*

The highest ranked process was the effect that the wind shadow has on the resuspension of particles, however, the resuspension rate itself was the lowest ranked process (Table 2.4). The diffusivity is the next most highly ranked process. If the seeding location does not have high islands (i.e. the wind shadow process is not relevant), the degradation process is the next most influential process. The settling process was 4th in the ranking, and therefore had a moderate influence on the result of the modelling. The wind-drift process, though not as influential as the wind shadow process, still had an effect on the model result and it therefore ranked as the 5th most influential process (Table 2.4).

For most processes, there was high variance in the ranks (Table 2.4), with the exception of the resuspension being affected by the wind shadow parameter which had a rank range of only 2.

Table 2.4: *Average rank of each scenario and the rank of the process. The highest ranked scenario (rank 1) is the scenario that is most different from the standard run.*

Scenario	Median rank	range	Median process rank
No wind shadow affecting resuspension	1	0	1
Diffusivity 5 m^2s^{-1}	3.5	4	4.25
Diffusivity 20 m^2s^{-1}	5	3	
Wind shadow 1000	9	9	8.75
Wind shadow 5000	8.5	11	
Macro- Settling 0.1	11.25	12.5	10
Macro- Settling 0.4	8	13	
Micro Settling 0.1	13	7.5	
Micro Settling 0.4	8.75	11.5	
Wind-drift 1%	10.75	11	10.38
Wind-drift 4%	10	11.5	
Macro- Resuspension 0.1	15.75	8	13.88
Macro- Resuspension 0.4	14	12	
Micro Resuspension 0.1	13.75	7.5	
Micro Resuspension 0.4	12	11.5	

2.4 Discussion

A number of processes determined the fate of modelled plastics. The most important parameter was the location of the source; a difference of just a few kilometres in a rugged topography made a very large difference to the fate of the plastics and, in particular, had a major influence on which beaches the plastics accumulated on and on the location and size of the plumes of suspended plastics. When the source was located near the coast and downwind, some of the plastics rapidly beached and the rest were rapidly advected away and spread vast distances. When the source was located in open waters the plastic plumes were more uniform and resembled the pattern expected from passive plumes emanating from a point source in uniform flows (Okubo, 1971). When the source was close to the coast and windward, the vast majority of the plastics beached quickly and the rest largely remained in coastal waters near the source. When comparing the known hotspot locations in the Whitsunday region to beaches that accumulate simulated plastics, there is a tentative relationship (Figure 2. 3), especially for simulations seeded from Site 4 and Site 1. It is important to note that these accumulation areas are very sensitive to seeding location (source of plastics) and not all source locations are known or have been modelled here. This study thus suggests that the seeding location had the largest influence on where debris accumulates on the coastline. In practice, the source location and the quantity of plastic debris is associated with a large amount of uncertainty in the literature (Reisser et al., 2013; Thiel et al., 2013), and thus clearly this should become a priority research area. Indeed, knowing the source location and the quantity of plastic debris is the prerequisite to using the model to quantify where plastics debris will accumulate for given wind and oceanographic conditions. In turn, this would lead to prioritisation of beach clean-up actions to minimise the cost and maximise the efficiency of combating plastic pollution. This would be the case especially for beached plastic pollution from urban rivers, or other land-based sources.

If a plastic pollution model was implemented in practice to guide management and government agencies, the next issue to solve would be parameterising accurately the resuspension of beached plastics by wind and waves in the presence of a complex topography on the mainland and the islands. This is the wind shadow effect proposed by Myksvoll et al., (2012) to explain the currents nearshore of a rugged coastline being different to those just offshore. This wind shadow effect also influences the resuspension of beached plastics. For plastics, this process is well known and was readily observed visually by the author (KC; unpublished data) but has not yet been quantified, and this should be the next priority research topic.

In terms of microplastics, the rate of degradation in ocean and beached environments will be essential in predicting the areas that will accumulate microplastics. As degradation is such a slow process it is likely that the sources of secondary microplastics are far away from where they are being found (Figure 2.5).

The turbulent diffusivity is a numerical process that is used in modelling to parameterise the movements within a single cell of the simulation grid that the model cannot capture (Okubo, 1971; Hrycik et al., 2013). As illustrated here, this parameterization has a determinant impact on the modelling result. A consensus on the best estimate of this coefficient in a rugged, shallow, coastal bathymetry remains lacking. Specific studies are required to obtain a reliable parameterization adequate to model debris transport in this region.

The other processes sketched in Figure 2.1 and described in Table 2.1 also contribute to determining the fate of plastics, and again most of them can be observed visually but have not been quantified. These should then be the next priority research topics. For instance, the rate of degradation of particles has more of an effect on the model results than the settling of particles. Also, the rate of degradation of macroplastics to microplastics on the beach has much smaller effect than the rate of degradation at sea. Again, the rate at which this occurs is not well understood (O'Brine and Thompson, 2010) and each polymer type has different physical properties making the rate slower or faster. The sensitivity analysis carried out in this study showed that the rate of plastic break-up in the sea and on beaches also helps determine the fate of plastics in estuaries and coastal waters. Another suggested priority research topic is thus the degradation rate of the most common polymers on beaches and in estuarine and coastal waters. The relative importance of these parameters may vary in areas with dramatically different topography, for example in areas with a very steep coast leading to a very deep continental shelf where the rate of settling may be more important.

Debris movements rely on the wind, however, the wind is variable in time. In this study, the effect of seeding time of the plastics (the date and time, relative to the weather that they enter the ocean) has not been explored, however, there is discussion of this process in Critchell et al., (2015). The seeding date, and therefore different weather conditions, does not seem to affect the location of the areas of accumulation of the coastline. This suggests that it is not as important to the modelling result as the parameters of wind shadow and the degradation of the plastics. Real wind and tide data are of course preferable when modelling the coastal system, however, it is well understood in the field of hydrodynamic modelling and incorporated into most models and is therefore not a priority for research.

Large amounts of the debris that accumulates on beaches are buoyant and are therefore directly influenced by the wind via the wind-drift coefficient (Daniel et al., 2002; Hardesty and Wilcox, 2011). This study suggests that the wind-drift coefficient is indeed an important process in the movement of buoyant marine debris. Most research that includes this parameter uses a figure between 1% and 6% (Yoon et al., 2010; Ebbesmeyer et al., 2011; Hardesty and Wilcox, 2011; Carson et al., 2013; Critchell et al., 2015; Maximenko et al., 2015), however, an accurate value is currently unknown. Determining the wind-drift coefficient of macroplastics should thus also be a research priority topic. In the same manner that determining the wind-drift coefficient of oil slicks was a priority research topic when oil spill models were first proposed and used operationally in the 1980s (Spaulding, 1988), there is much similarity between the original developments in research of oil slick modelling thirty years ago and plastic pollution research that is just starting to emerge.

The method of modelling used here to predict the short-term fate of plastics can readily be applied to any shallow estuarine and coastal waters. All that is needed is a reliable, high-resolution oceanographic model, the knowledge of the local wind field and the influence of the local topography on that wind. An advection diffusion model for plastics can then be added to the oceanographic model that includes all the processes shown in Figure 2.1. The incorporation of the wind shadow effect is simple in this method; in some cases one may need accurate small-scale atmospheric models to better solve the wind field around islands and a rugged coastline. In the shallow, vertically well-mixed coastal waters of the study area, the currents produced by the two methods vary very little with depth and a 2-D model is appropriate, as justified by Critchell et al., (2015); however, in stratified waters, a 3-D model would be needed.

Although many plastic items we use are made of buoyant polymers, biofouling and inundation of whole items can cause them to sink to the benthic environment (Chubarenko et al., 2018). The rate at which this occurs is little known. Clearly research is needed on the physical properties of plastics in the environment, and is important for modelling their fate in estuarine and coastal waters. The modelling of the resuspension of beached plastics is also simple in the method used here; it is parameterised by a rate, itself a function of the wind shadow. Again, there are no data on this process; presumably this resuspension may be better parameterised in the future by including a model of the wave field along a rugged coastline and the effect of the waves on the resuspension. This is another suggested priority research area. The implementation of the settling parameter in the model presented is simplistic at this time and there is no movement once the particles settle to the seabed. This should be improved once the other processes are better understood.

In summary, the study suggests that the processes that most influence the accumulation of plastics on beaches are, in decreasing order, the source locations and quantities of debris, the degradation of macroplastics into microplastics at sea and on beaches, the resuspension of beached plastics in relation to the wind shadow effect, the diffusivity, the wind-drift coefficient of floating plastics, and the rate at which plastics sink. I suggest a focus on laboratory and field studies in these areas to increase knowledge and to rapidly improve the reliability of marine debris models. In practice, different plastics (e.g. plastic bags, plastic bottles, bottle caps, fishing nets, buoys and lines, microbeads used in cosmetic and personal care products such as facial exfoliators, body scrubs and toothpastes, etc.) are made from different polymers that have different physical properties and, in turn, these may vary the relative importance of the various physical processes controlling the fate of plastic debris in estuaries and coastal waters. Thus, ultimately the plastic oceanography model may need to evolve from using unique values of the different parameters characterising these processes (see Table 2.1) and instead simultaneously use a range of values of all these parameters in order to represent the wide range of plastic debris.

Chapter 3
Predicting the exposure of coastal habitats to plastic pollution

In Chapter 2, I presented a new plastic-specific hydrodynamic model. In this chapter, I use the model to predict the potential exposure of vulnerable habitats and species, namely mangroves, coral reefs and marine turtles to plastic pollution. The effect of plastics on the marine environment is well documented, however, the physical locations of these interactions remain unknown. I assessed the potential exposure of mangroves, coral reefs and marine turtles to plastics during the two main wind conditions of the region; the trade winds and Monsoon wind seasons. I found that, in the trade wind season (April to September), reefs, mangroves and turtles all had lower exposure than during the Monsoon wind season (October to March). A small proportion of coral reef habitat was in the high exposure categories, whereas a large area of turtle habitat was in high exposure categories. Unlike reefs and turtles, the mangrove habitat had consistent hotspots of high exposure across wind seasons. The outputs of this chapter inform local-scale management action, for example turtle management and recovery plans. The method presented here can also be transferred to other species and habitats and scaled up for larger jurisdictions.

Citation: Critchell, K. Hamann, M., Wildermann, N., and Grech, A. in prep "Predicting the exposure of coastal habitats to plastic pollution" target journal: Biological Conservation

3.1 Introduction

The allocation of limited conservation resources is a difficult task for environmental managers (e.g., Fuentes et al., 2015). To be effective at solving environmental challenges, priorities and resources need to be allocated between all types of environmental management activities or intervention: remediation, rehabilitation, restoration, and active management (Gilbert, 2011; Game et al., 2013). These activities would ideally be prioritised and implemented in areas where they are expected to have the greatest benefit and are most likely to succeed at stabilising or improving the state of the environment. For each environmental issue, setting priorities and determining the areas for resource input is challenging because economic conditions, uncertainties in the state or condition of the environmental asset, ecosystem services or ecosystem value are often not well quantified or understood, and all need to be considered when choosing areas to allocate resources (Margules and Pressey, 2000). Prioritisation tools, such as risk assessments, and structured decision making are used to objectively assess, and compare expected success, of allocating resources towards conservation activities (Wilson et al., 2006; Klein et al., 2017).

In conservation or environmental management contexts, risk assessments are used to inform decision making about resource allocation or mitigation of pressures. In particular, risk assessments can be used to assist prioritization by identifying areas on a scale from low to high risk from threatening activities. Knowing this enables strategic decisions for management action to deliver cost-effective benefits. In some cases, managing the low risks are likely to succeed, whereas areas at high risk may be discounted as lost causes (e.g. Bottrill et al., 2008). In other situations it may be prudent to address only the higher risks in order to remove stressors and to mitigate against further harm (Margules and Pressey, 2000; Halpern et al., 2006). Risk assessments have two measurable components: the likelihood of a hazard event occurring and the consequence to a value should the hazard event occur. In the context of environmental risk assessments, a hazard event or threat is any process or action that can affect the health or condition of an environmental value (Norton et al., 1996; Grech and Marsh, 2008). Consequence is the impact or damage that occurs, directly, or indirectly, due to the threat. A risk assessment can be visualised using a spatial risk assessment, which is a spatial depiction of the risk analysis (reviewed by Lahr and Kooistra, 2010). For example, Grech et al., (2011) assessed the cumulative impacts of anthropogenic threats to coastal seagrass meadows, finding urban and port developments to be major contributors to the risk to seagrass meadows.

Information on the spatial distributions of threatening processes is integral to informing the development of spatial risk assessments. Modelling is often used to predict the spatial distributions of species and threats (e.g. Grech and Marsh, 2008; Halpern et al., 2008; McPherson et al., 2008; Grech et al., 2011; Halpern et al., 2015), and is especially useful in the marine environment where data are sparse, and expensive to obtain, relative to many datasets collected in the terrestrial or freshwater environments (Ban, 2009; Brown et al., 2011). In particular, ecological niche-based models and species dispersal models are widely used to determine distribution of wide-ranging marine species such as marine turtles and sharks (McKinney et al., 2012; Wildermann et al., 2017), and hydrodynamic modelling is commonly used to understand the way the threats, such as water pollution, are distributed and diluted away from the source (Li et al., 2000; Cucco and Daniel, 2016). In a more applied setting, modelling has been used in a variety of risk analysis studies of marine systems, such as to determine the spatial extent, or likelihood of threat exposure (e.g. industry and marine turtles Whittock et al., 2016). Modelling has been used to assess the risk of plastic pollution (e.g. Wilcox et al., 2013; Wilcox et al., 2015), however, these studies are conducted in large areas and at relatively coarse spatial resolution. To inform management at a fine scale (e.g. the Whitsundays Special Management Area (GBRMPA): 1688 km^2; Figure 3.1) these studies would not be suitable, as the resolution is inadequate.

As discussed in Chapter 1, plastic pollution is emerging as a threat to the marine environment. The negative consequence of exposure to plastic for a variety of marine species is only beginning to be understood, especially regarding the population-scale effects of ingestion (Worm et al., 2017). For example, marine turtles ingest small plastic particles, causing disruption to their gastrointestinal tract (Di Bello et al., 2013; Colferai et al., 2017), but most data are presented at an individual scale and not at management-related scales such as foraging areas. Sensitive habitats, such as mangroves and coral reefs, can be damaged by scouring or smothering by larger plastic items (e.g. Uneputty and Evans, 1997; Donohue et al., 2001). However, the spatial location of where these interactions occur, and the frequency of interactions, especially at a management-relevant scale, are not well known (Titmus and Hyrenbach, 2011; Nelms et al., 2016).

The goal of this chapter is to present a method of predicting the spatial distribution of exposure of coral reef systems, mangrove habitats, and foraging flatback sea turtles to plastic pollution in a complex coastal environment, namely the Whitsunday region, Queensland, Australia (Figure 3.1). I used hydrodynamic modelling to estimate distributions of plastics and to create exposure categories. The exposure categories were then compared with known distributions of threatened species or habitats. The outputs of this chapter improve the understanding of the risks of plastic pollution as well as providing a tool to improve the management of plastic pollution in the coastal zone.

Table 3.1: *Proven and speculated consequences of exposure to plastic pollution for each of the study habitats and species.*

Habitat or species	Consequence	Macro- or microplastics	Speculated or evidence of consequence	Exposure layers used in analysis
Coral Reefs	Scouring/smothering corals	Macro	**Speculated** (Goldberg, 1997; Donohue et al., 2001)	Settled macro
	Ingestion by animals in the habitat	Micro	**Evidence** (Gall and Thompson, 2015; Hall et al., 2015)	Settled micro
	Invasive species	Macro- and micro	**Evidence** (Barnes, 2002; Gregory, 2009)	Settled macro- and micro
Turtles	Gastrointestinal disruption after consumption	Macro- and micro	**Evidence** (Parga, 2012; Di Bello et al., 2013; Schuyler et al., 2014; Nelms et al., 2016)	Suspended macro- and micro
	Entanglement	Macro	**Evidence** (Wilcox et al., 2013; Nelms et al., 2016; Blasi and Mattei, 2017; Duncan et al., 2017)	Suspended macro
Mangroves	Scouring/smothering	Macro	**Speculated** (Goldberg, 1997; Uneputty and Evans, 1997; Smith, 2012)	Beached macro
	Changed community structure	Macro	**Evidence** (Katsanevakis et al., 2007)	
	Ingestion by animals in the habitat	Micro	**Evidence** (Besseling et al., 2013; Gall and Thompson, 2015; van Cauwenberghe et al., 2015)	Beached micro
	Degradation of habitat	Micro	**Speculated** (do Sul et al., 2014)	
	Invasive species	Macro- and micro	**Evidence** (Barnes, 2002; Gregory, 2009)	Beached macro- and micro

3.2 Methods

3.2.1 Study area and species

This study was conducted in the Whitsunday region of the Great Barrier Reef World Heritage Area (GBRWHA), on the central coast of Queensland, Australia (Figure 3.1). The largest township is Airlie Beach which has a large transient tourist population and is the centre for local government activities. The average water depth in the region is approximately 30 metres, and the region is dotted with 77 islands and reefs. The Whitsunday region is towards the southerly extreme of the tropics, dominated by the monsoonal and south-easterly trade wind circulations. During the monsoonal summer months, the Whitsundays receives run-off from three mainland catchments; The Don, Proserpine and O'Connell catchments (Figure 3.1). These catchments are predominantly of agricultural land-use.

There is growing evidence that sea turtles are at risk from plastic exposure (e.g. Schuyler et al., 2012; Schuyler et al., 2014). Flatback turtles (*Natator depressus*), endemic to Australia, are listed as "Vulnerable" under the Environmental Protection and Biodiversity Conservation Act 1999 (EPBC Act), and are an important value of the GBRWHA. In addition, plastic debris is recognised as a threat to flatback turtles residing in the GBRWHA and elsewhere in Australia (summarised in Table 3.1). Understanding the threat posed by plastics to this species is important and can also be extrapolated to other pelagic and sea turtle species. Indeed, Nelms et al., (2016) and Darmon et al., (2017) call for research on the risk of plastic pollution for marine turtles, especially flatback turtles, and research on foraging and threats to flatback turtles is a priority under the GBRMPA's Reef 2050 Plan (Reef 2050 Long-Term Sustainability Plan , Commonwealth of Australia 2015). Approximately 19 turtles (species information not available) per year were stranded in the Whitsundays region between 2005 and 2010 and most of the turtles had no visible signs of boat strike or entanglement (Biddle and Limpus, 2011). This trend continued until 2016 (unpublished data, StrandNet). Four out of the five Flatback turtles that could be necropsied between 2013 and 2017 had microplastic blockage in their gastrointestinal tract (unpublished data, StrandNet). The Whitsundays region is one of the most important foraging areas for the flatback turtle population breeding in the Great Barrier Reef (Wildermann, 2017)

Coral reefs are one of the key environmental values of the GBRWHA and they are a hotspot for biodiversity, not only making them biologically important to the health of the ocean but also important to the Whitsunday region specifically due to their notoriety for tourism. Mangrove habitats are important nursery grounds for many commercially important species, as well as providing many ecosystem services. Understanding the impact of plastics on these habitats can lead to benefits for all species living within, or reliant on, mangroves and coral reefs – especially in the GBRWHA.

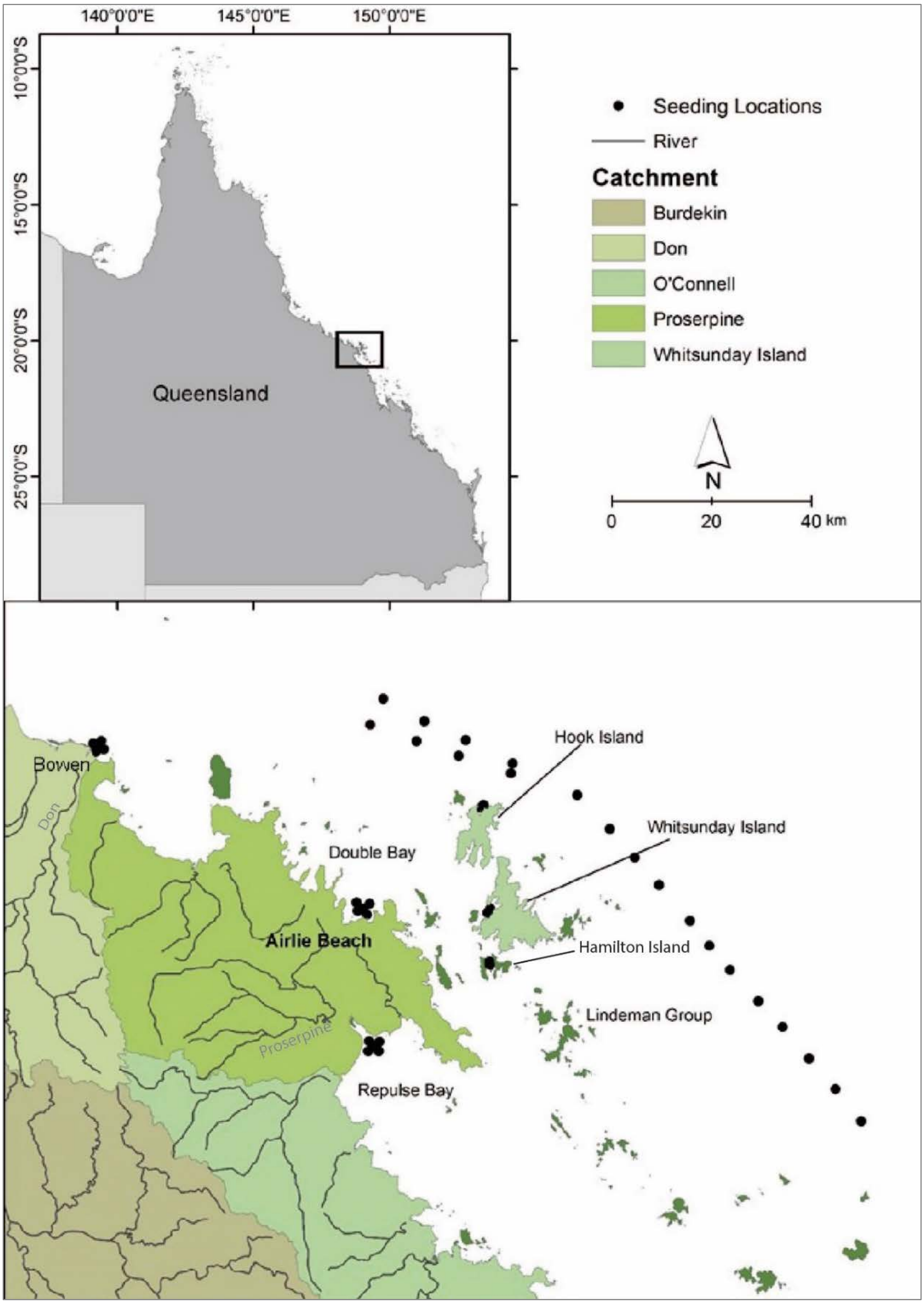

Figure 3.1: The Whitsunday region. The lower panel shows the placement of the hydrodynamic simulation seeding locations, shown as black circles. The river catchments are shown in green hues, with streams and rivers shown in grey. The top panel shows the placement of the region on the Queensland coast.

3.2.2 Hydrodynamic modelling and dispersal simulations

I used the SLIM hydrodynamic model (described in Chapter 2) to model the dispersal of plastic pollution in the study area. Data inputs included recorded wind data collected at Shute Harbour (Bureau of Meteorology station number 33106) from June 2013 to May 2014. This enabled the comparison of plastic pollution distribution during the south-east trade wind season (April - September) and during the more wind-variable northerly season (hereafter, referred to as "Monsoon wind season"; October - March). It is important to compare these seasons as the effect of wind determines the movements of marine plastics (See Chapter 2: Sensitivity Analysis), and the two wind seasons capture the maximum variability during the year. The years 2013 and 2014 provide good examples of the conditions during the two wind seasons, showing average wind distribution patterns for the seasons (Figure 3.2).

I imposed a constant wind shadow in the lee of islands of 2500 m. In reality, the size of the wind shadow would change with the size and shape of the land mass causing it. However, to implement a variable wind shadow in the model would require coupling a wind-field model to the SLIM model. This was outside the scope of my thesis as it would require substantial modification to the course code of the SLIM model. The model was forced with a standard M2 tide inflow and forcing from the Coral Sea; both are idealised but have been successfully used in previous studies to provide an acceptable representation of water movements (Hamann et al., 2011; Andutta et al., 2013; Critchell et al., 2015).

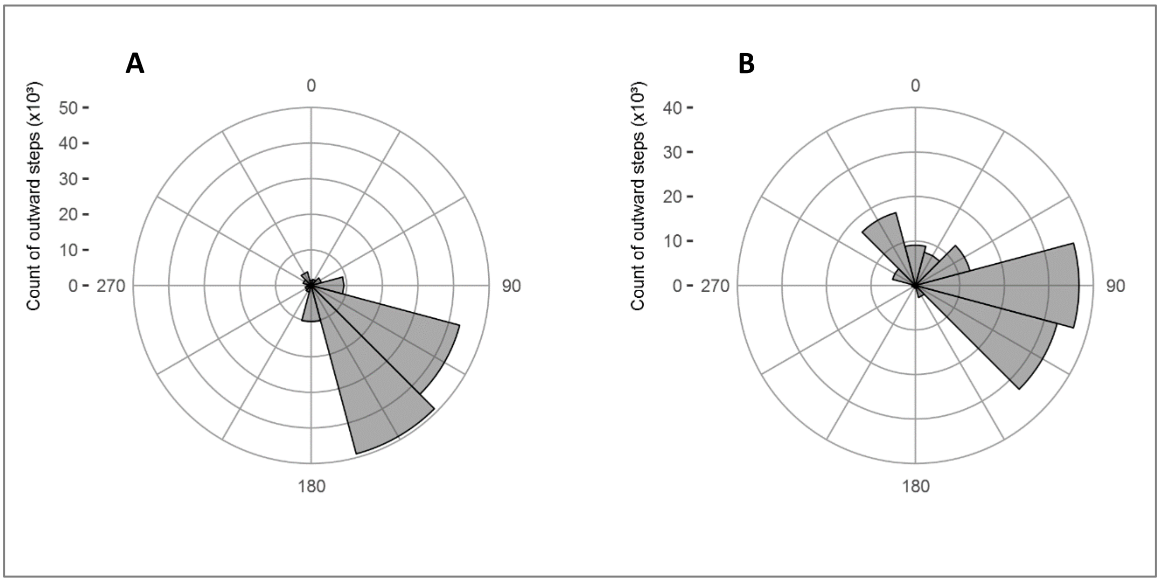

Figure 3.2: *Wind rose of the wind data used in the modelling that created the exposure layers for each season, A) trade wind season and B) Monsoon wind season. Wind data from 10 min wind records at Shute Harbour weather station.*

I used the most likely sources of plastic pollution for the study area, as per Chapter 2, as the starting locations for the particles to 'seed' the model (Figure 3.1 and Table 3.2). I released 250,000 particles at each source point, therefore, locations with more source points were considered of higher importance (Table 3.2). Land/catchment based sources included the major river systems flowing into the study region. These catchments have a multitude of land uses, including urban and agriculture, and the likelihood of them contributing plastics is high (Moore et al., 2011; Critchell et al., 2015). I also chose the waterbodies that drained any local water treatment facilities as they are a likely source of microplastics (Browne et al., 2007; Fendall and Sewell, 2009). Hamilton Island resort has their own water-treatment facility, and a large tourist turn over. I included Hamilton Island as a source of both macro- (tourist and resort-based litter) and microplastics from the water treatment plant (e.g. fibres fabric washing daily). Offshore sources were evenly spaced along the offshore commercial shipping lane (Chapter 2). I did this to include the shipping lane itself as a source, but also as a surrogate for other external sources, as many offshore sources are diffuse and therefore the appropriate seeding location is not obvious.

Table 3.2: *The justification and importance of each source location in the plastic dispersal simulations.*

Location	Rank of importance	Number seeding points	Source justification
Airlie Beach	High	5	Regionally large (population 12928; Census, 2016) local township
Mouth of the Proserpine River	High	5	Large catchment draining many land uses
West Hamilton Island	Low	2	Popular resort with on-site water treatment
Cid Harbour - West Whitsunday Island	Low	2	Large tourist anchorage
North Hook Island	low	2	Large tourist anchorage
Bowen	High	5	Regionally large (population 9105; Census, 2016) local township
Outer Shipping lane	High	20 evenly spread	Objects discarded by ships, also ship sewage

The dispersion model uses the velocities derived from the hydrodynamic model to move particles according to the Lagrangian dispersion scheme. Within this I also added plastic-specific parameters (described in Chapter 2; Critchell and Lambrechts, 2016). To understand the distribution of plastics in the Whitsundays region, I used the best estimation of these parameters, informed by available literature. I set the wind-drift to 2% of the wind velocity for simulated macroplastic particles and zero for microplastics. The resuspension (or re-floating) probability was set to 0.2 for both macro- and micro- particles. I assume the resuspension is mainly wave driven, therefore, the resuspension is turned off in the areas designated as being in a wind shadow. The wind shadow was set to the same value as for the hydrodynamics model. The rate of settling is set as a probability, 0.002 for macroplastics and 0.02 for microplastics, as microplastics are more likely to become bio-fouled causing sinking, or to be flocculated into marine snow. The rate at which macroplastics "degrade" into microplastics is 0.000001 for particles in suspension and 0.00001 for particles on the land. Plastics on land are thought to have a higher rate of degradation, especially in tropical regions, because they are exposed to higher UV intensity and temperatures that degrade polymer bonds (Weinstein et al., 2016). However, the process of full decay is thought to be at the scale of months to years and this rate may be still too fast. The simulations were run for a maximum of 45 days and three outputs (days 15, 30 and 45) were exported to capture variability in dispersal throughout the simulation length. I ran one simulation for each wind season, the trade wind season simulation began on 01/06/14 and ran for 60 days, and the Monsoon wind season simulation started 01/02/14 and ran for 45 days.

3.2.3 Exposure layers

The outputs of the two simulations were imported to ArcGIS 10.2. The outputs were used to create macroplastic and microplastic exposure layers for each wind season (trade and Monsoon) and each particle state (suspended particles, beached particles and settled particles; n = 12). The methods to create the exposure layers are shown graphically in Figure 3.3.

3.2.3.1 Beached particles

I used the spatial join function to join particles to their closest coastline section on days 15, 30 and 45 of the simulation. I merged the outputs of days 15, 30 and 45 to create an exposure layer of the mean particle density. Mangrove habitat tends to be coastal, therefore, I compared mangrove presence to the beached plastic exposure layers. I first joined the polyline of mangrove presence to the SLIM coastline in a binary format (0, 1). I then multiplied the mean particle density by the habitat presence to assess the relative exposure of each 100 m section of mangrove habitat.

3.2.3.2 Suspended particles

I created a density distribution, with the Kernel Density function of ArcGIS for each macro- and microplastic output at 15, 30 and 45 days. The kernel density function creates a raster grid by counting the number of points in each of the cells and smoothing through the values of the surrounding cells in a defined search radius. I used the default search radius, calculated by the programme using Silverman's Rule of Thumb for each of the input datasets. The result is a surface where each cell value is the sum of the values created by each search. I then used the mean concentration of these outputs (n = 3) to create the exposure layer for suspended macro- and microplastics.

3.2.3.3 Settled particles

Settled particles in the model represent the accumulation of settled particles throughout the simulation. Therefore, the output on day 45 provides the locations of all particles that have settled on the ocean floor. I used the last output (day 45) to create the exposure layers for settled plastics, using the same Kernel Density method as for the suspended particles.

3.2.4 Relative exposure categories

The layers of beached, suspended and settled macro- and microplastics during the trade and Monsoon wind seasons were divided into four “Relative Exposure Categories” (RE categories). The RE categories were: Nil (where plastics are not present), low, medium, and high. The layers of each particle type (macro- and micro) and state (beached, suspended and settled) had different frequency distributions of particle density. It was therefore necessary to develop these categories individually, but retaining the ability to compare the trade vs. Monsoon wind layers (e.g. suspended macro- plastics in the trade vs. Monsoon wind season). The breaks used for the categories were based on the quantile distribution, which is an inbuilt classification method to break the data into classes with an even number of values within each class.

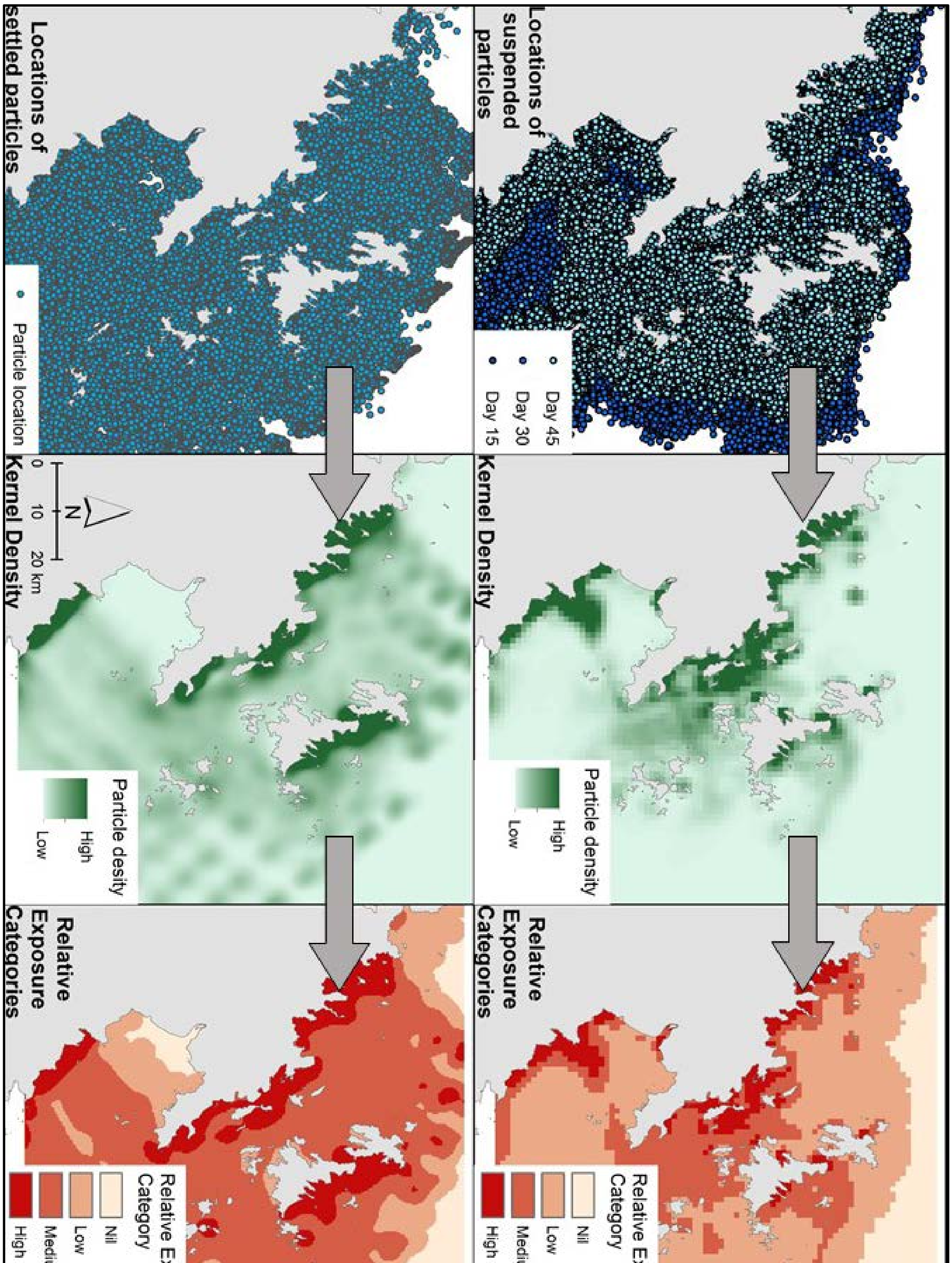

Figure 3.3: *Example of the process used to create the relative exposure categories. The top panels show the process for suspended particles, whereas the bottom panels show the process for particles that have settled. The particles in this example are macroplastics in the Monsoon wind season. The particle locations were converted to a continuous grid via kernel density and then the quantile breaks used to split the data into four categories.*

3.2.5 Habitat and organism distribution data

Wildermann (2017) found that the Whitsundays region is one of the important habitats for flatback turtles breeding in the Great Barrier Reef. To assess the exposure of flatback turtles to plastics, I used the location data derived from 13 adult female turtles tracked in the Whitsunday region by satellite-linked Fastloc GPS transmitters (Wildlife Computers) between December 2009 and December 2015. The data were used to model the spatial distribution of turtle foraging areas. Each of the turtles was fitted with the transmitters after they laid their last clutch in a breeding season and they were then tracked to their foraging areas (e.g. Wildermann, 2017). Following arrival at the foraging site, the tags transmitted location data for a mean of 171.2 days. The x,y coordinate data were extracted and screened with a data-driven filter (Shimada et al., 2012) to remove temporal and spatial duplicates, and locations marked by biologically unlikely swimming behaviour (> 7.6 km/h) and turning speeds (> 1.8 km/h). Home ranges were then estimated from utilisation distributions (i.e. all the area used by the animal) using the method of Calenge (2011). The 95% core home ranges were used to develop a binary turtle presence/absence raster layer (1 km x 1 km resolution; Figure 3.4) (e.g. Whittock et al., 2016). The geographic area used by these 13 turtles was assumed to overlap with suitable foraging habitat (Wildermann et al., 2017), and therefore the presence of these turtles can be used as a proxy for turtle foraging habitat in the Whitsunday region.

Reef and mangrove habitat location data were obtained from the Australian Institute of Marine Science eAtlas data depository (http://eatlas.org.au/). The reef data layer was converted into raster format (330 m x 330 m resolution) to enable its comparison with the RE layers. (Figure 3.4) The mangrove data is a polyline feature and was joined to the SLIM coastline with a spatial join, in order to label the SLIM (simplified) coastline layer with sections classed as mangrove habitat (Figure 3.4).

The interaction of turtles, reefs and mangroves with plastics in the three states (i.e. beached, suspended and settled) differs (*see* Study area and species; Table 3.1). Coral reef species are most affected by plastics that settle onto the reef matrix (settled). Turtles are most likely impacted by plastics suspended throughout the water column, and mangroves are a coastal habitat type likely to be affected by plastics that are pushed onto the coastline (beached). I matched turtles, reefs and mangroves to their relevant exposure layer when conducting the risk analysis.

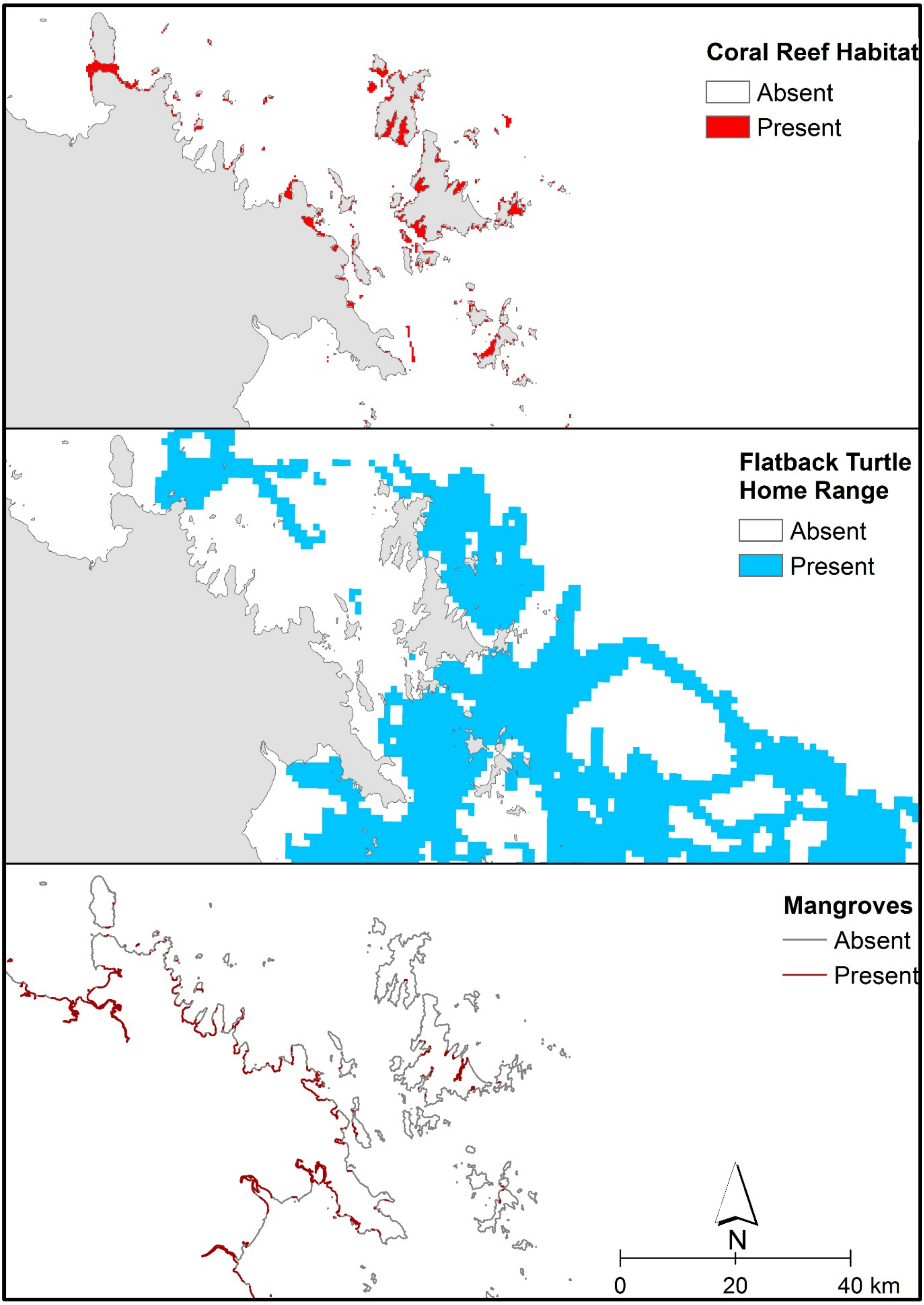

Figure 3.4: *Study area, the Whitsunday region. Top panel shows the locations of reef habitat in the study region, centre panel shows the home ranges of flatback turtles, and the bottom panel shows the coastline designated as Mangrove habitat.*

3.2.6 Data analysis

The exposure layers and the presence/absence habitat and species data were overlayed in GIS to calculate the area of mangrove, coral reef and turtle habitat in each RE category. I used a raster calculator to multiply the binary (0 or 1) habitat layers with the exposure layers, resulting in a new raster layer with only the area of the habitat retaining the risk category of the exposure layer. To calculate area of exposure in each category, I multiplied the number of cells in the category by the cell dimensions (330x330 m for the coral reef habitat and 1000x1000 m for the turtle home ranges). To ensure the exposure layers and the habitat layers covered the same spatial extent, I used a mask to extract only the data within the study area for analysis. For the interaction of plastics with mangroves, I calculated the length (total of 1307 km) of mangrove coastline in each threat category. I summed the lengths of the coast sections in each threat category that were also labelled as mangrove habitat.

3.3 Results

The trade wind and Monsoon wind seasons resulted in different spatial patterns of accumulation for, settled (Figure 3.5), suspended (Figure 3.7) and beached (Figure 3.9) macro- and microplastics. The trade wind season moved simulated macro- and microplastics into the large south-east facing bay at the southern end of the Whitsunday region (Repulse Bay). In comparison, the Monsoon wind season moved plastics into the smaller, more complex bays in the north of the study region (Double Bay etc.). The macroplastics accumulate close to the north-east facing coastlines during the Monsoon wind season, while microplastics had a more homogenous distribution. In the trade wind season, both plastic types had pockets of high accumulation (Figure 3.5). Microplastics accumulated on the beaches of the Lindeman group (south of the study region Figure 3.2) in the trade wind season; however, there is no such accumulation for the macroplastics (Figure 3.9).

The distances that particles moved also differed between seasons. During the trade wind season, the simulated microplastics moved a maximum of 619 km to the north, whereas macroplastics moved a maximum of 650 km north after 45 days of simulation. Few particles moved south of their original source location, 20.52% and <1% of simulated particles of microplastic and macroplastic, respectively. By contrast, during the Monsoon wind season the simulated microplastics moved only a maximum of 107 km to the north, and macroplastics 62 km. A relatively large proportion of particles moved south of their source location, 59.51% of microplastics, and 46.10% of macroplastic.

In the trade wind season, when winds are stronger, particles are pushed from the study area by wind-driven currents, resulting in the lower accumulation of plastics in the study area. It is possible that the plastics are accumulating in locations outside the study area. In the Monsoon wind season, particles remain in the study area trapped in bays moved by tidal currents, but also pushed by the wind against the coast, which leads to higher exposure in the Monsoon wind season.

The level of exposure of coral reefs, mangroves and marine turtles to macro- and micro plastics differed between wind seasons. A consistently higher proportion of each habitat was in higher RE categories in the Monsoon wind season compared with the trade wind season because simulated plastics were swept away from the islands as in the trade wind season (summarised in Figure 3.11).

Coral reefs had the largest proportion of area in the nil RE category, with a mean 58% of habitat across season and plastics type (Figure 3.6, 3.11). Coral reefs had low exposure during the trade wind season; for macroplastic 94.6% of the habitat, and for microplastics 99.8% of the habitat is in low or nil RE categories. Conversely, during the Monsoon winds season 31.7% of the habitat was in the medium or high RE category for macroplastics and 28.1% for microplastics (Figure 3.11). Only one cell (330 x 330 m) was consistently in the high RE category across wind season for microplastics and none for macroplastics (Figure 3.12).

Similarly, a large area of flatback turtle home ranges was in the medium and high RE category in the Monsoon wind season, compared with a small percentage during the trade wind season (for microplastics 51% and 1.7% respectively). The exposure was spread throughout the flatback turtle home range (Figure 3.8). Unlike the coral reef habitat however, there is a clear difference in exposure between macro- and microplastic. Specifically, in the Monsoon wind season 14.5% of the home range area was in medium or high RE categories for macroplastics, but 51.0% for microplastics (Figure 3.8, 3.12). A very small portion of flatback turtle home range (20 km^2, 0.59% of the total area) was consistently in the medium or high microplastic RE category between seasons, while only 2 km^2 of the area was consistently in medium and none in the high RE category for macroplastics (Figure 3.12).

The mangrove habitat had complex exposure patterns. Mangroves had the smallest proportion of its range in the nil RE category (mean 19.8%; Figure 3.10, 3.11) relative to marine turtle habitats and coral reefs. Unlike the coral reef habitat and the turtle home range exposures, the proportions of mangrove habitat in each RE category is reasonably consistent across wind season and plastic type. Also, unlike the other two case studies, for mangroves there are geographic areas that remain in the high RE category, suggesting there are consistent hotspots of exposure in time and space. For example, much of the mangrove habitat in Pioneer Bay, surrounding Airlie Beach, is consistently in the high RE category (Figure 3.12).

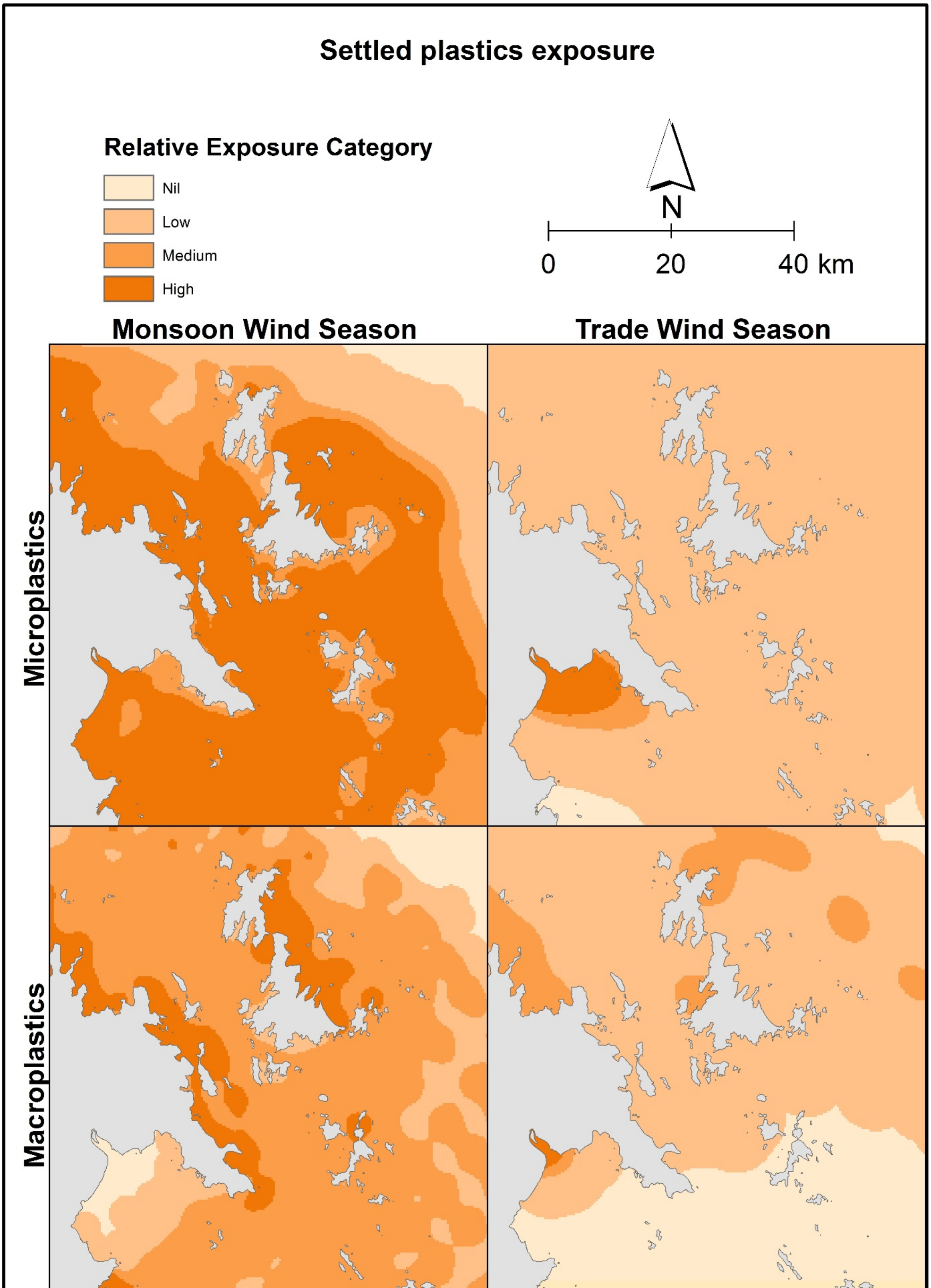

Figure 3.5: *The spatial distribution of relative exposure for settled macro- and micro plastics during the Monsoon (October - March) and trade wind (April - September) seasons.*

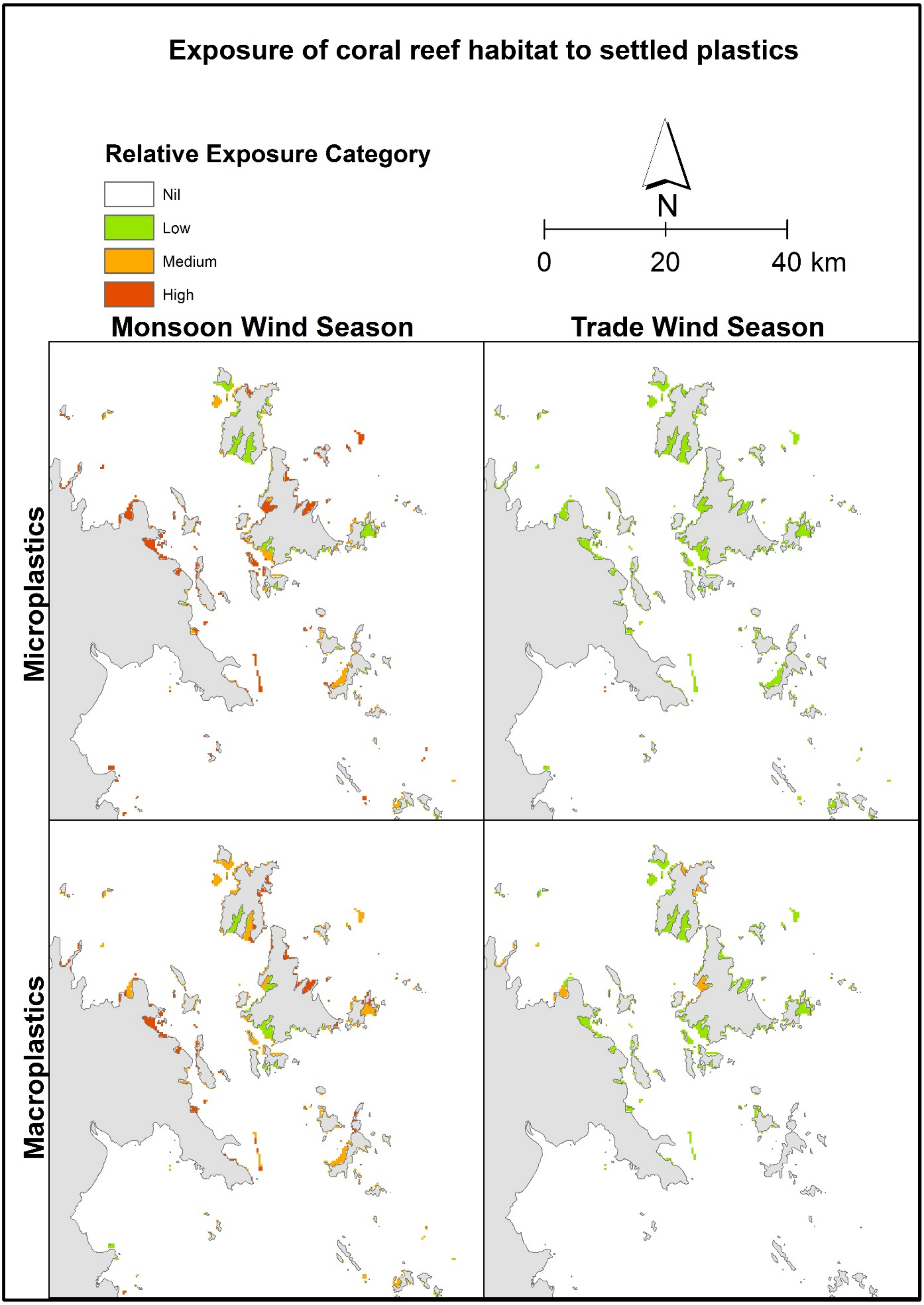

Figure 3.6: *The relative exposure to macro- and micro plastics for reef habitats in the Whitsunday region*

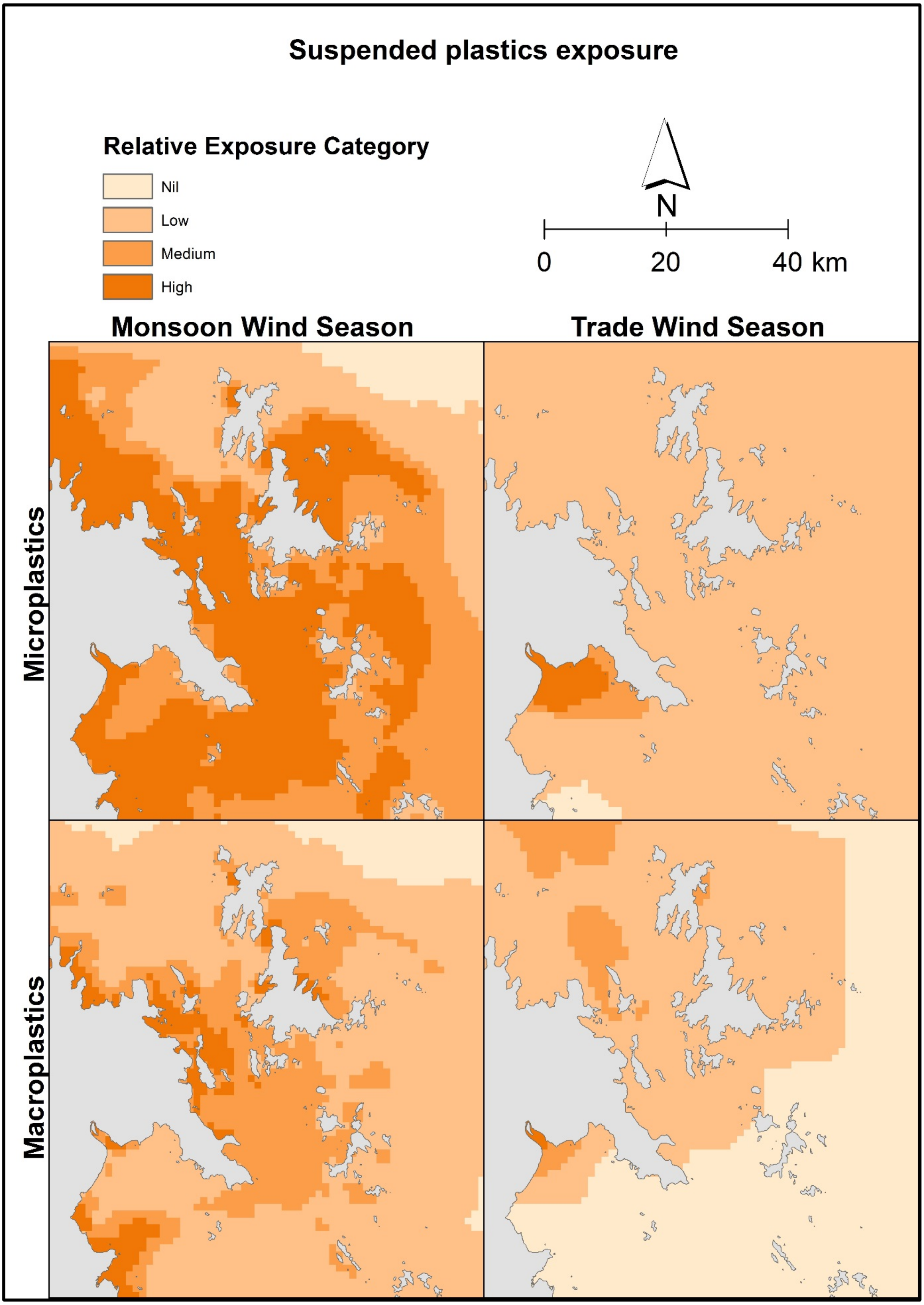

Figure 3.7: *The spatial distribution of relative exposure for suspended macro- and micro plastics during the Monsoon (October - March) and trade wind (April - September) seasons.*

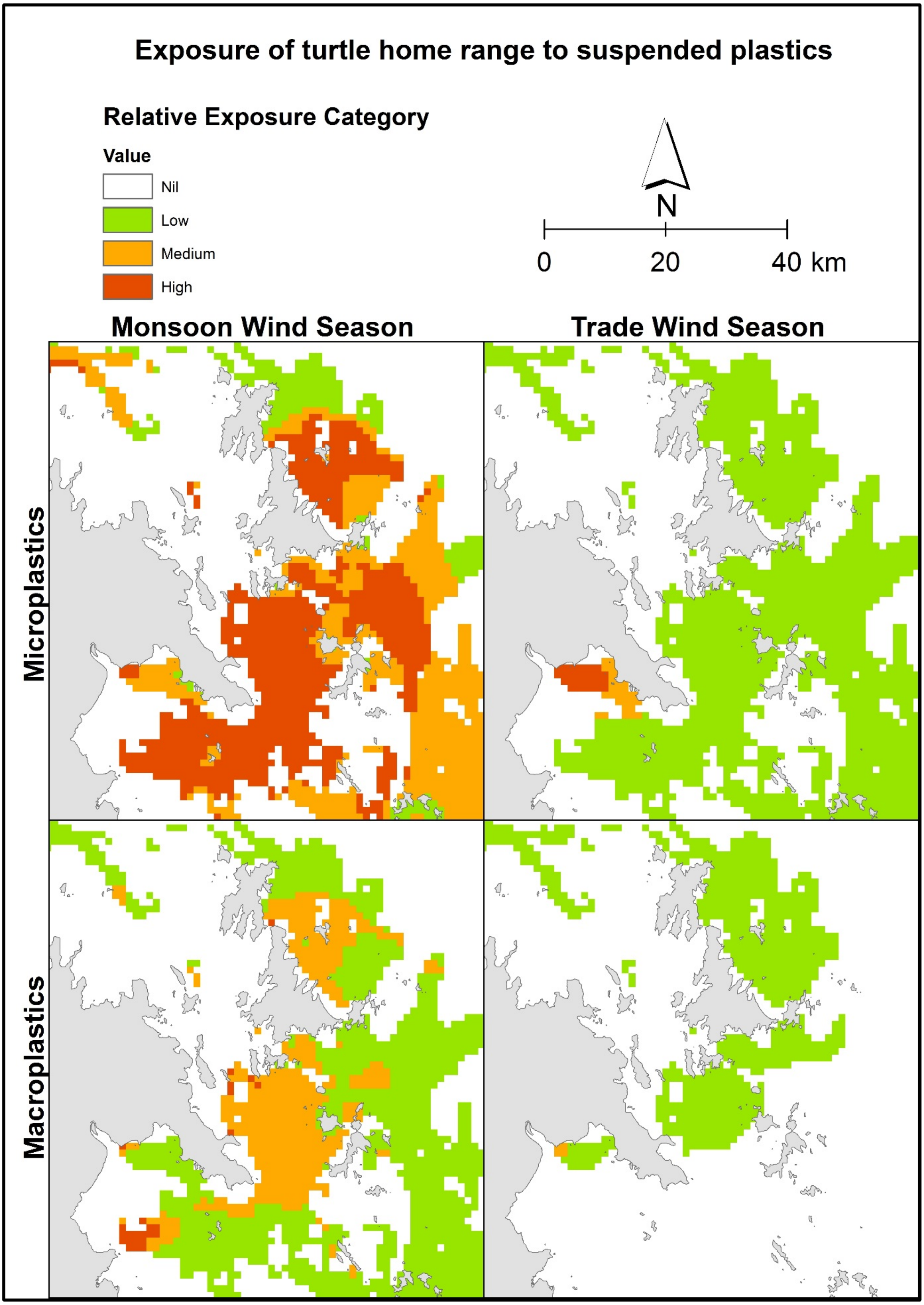

Figure 3.8: *The relative exposure to macro- and micro plastics for flatback turtle home ranges in the Whitsunday region*

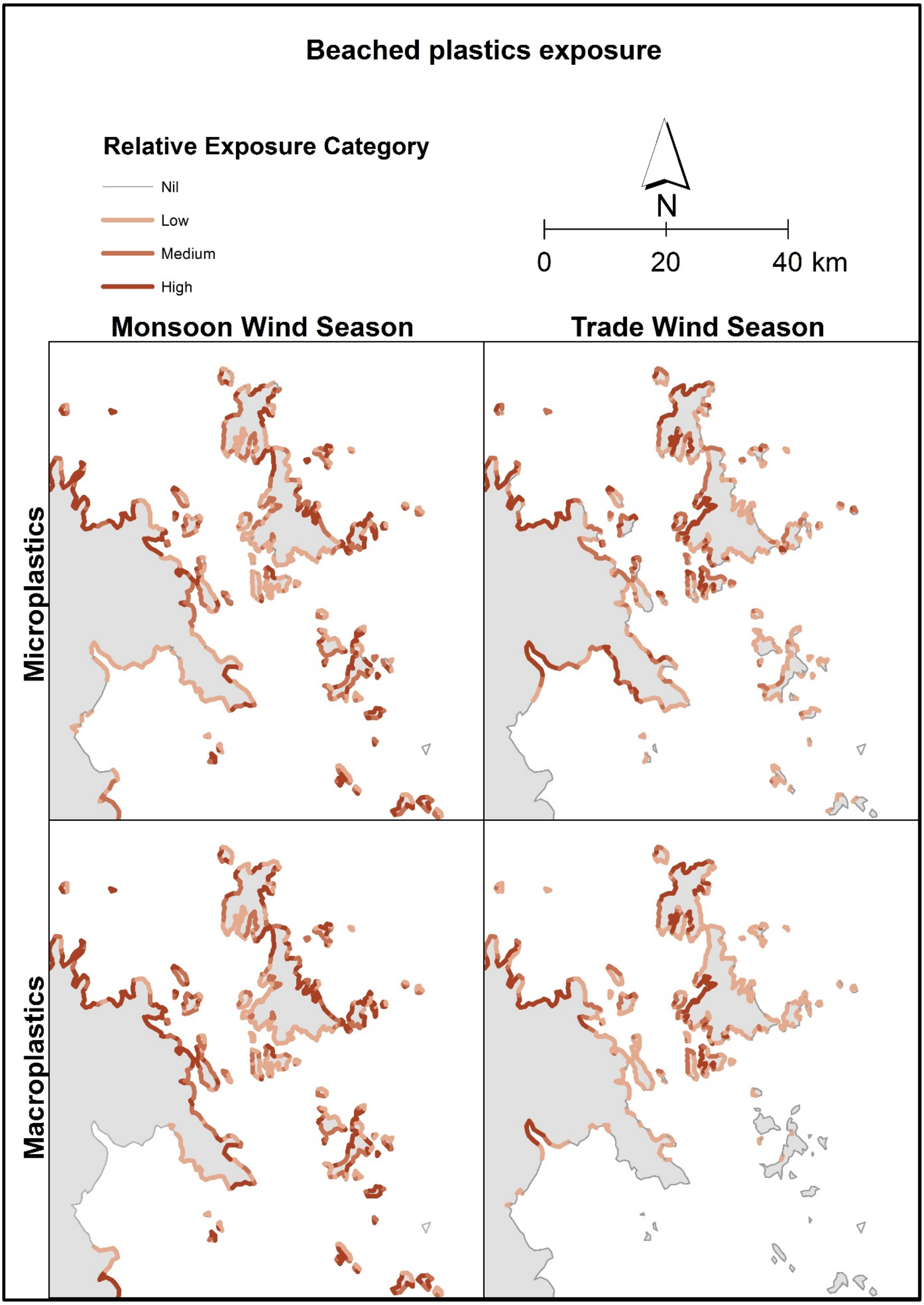

Figure 3.9: *The spatial distribution of relative exposure for beached macro- and micro plastics during the Monsoon (October - March) and trade wind (April - September) seasons.*

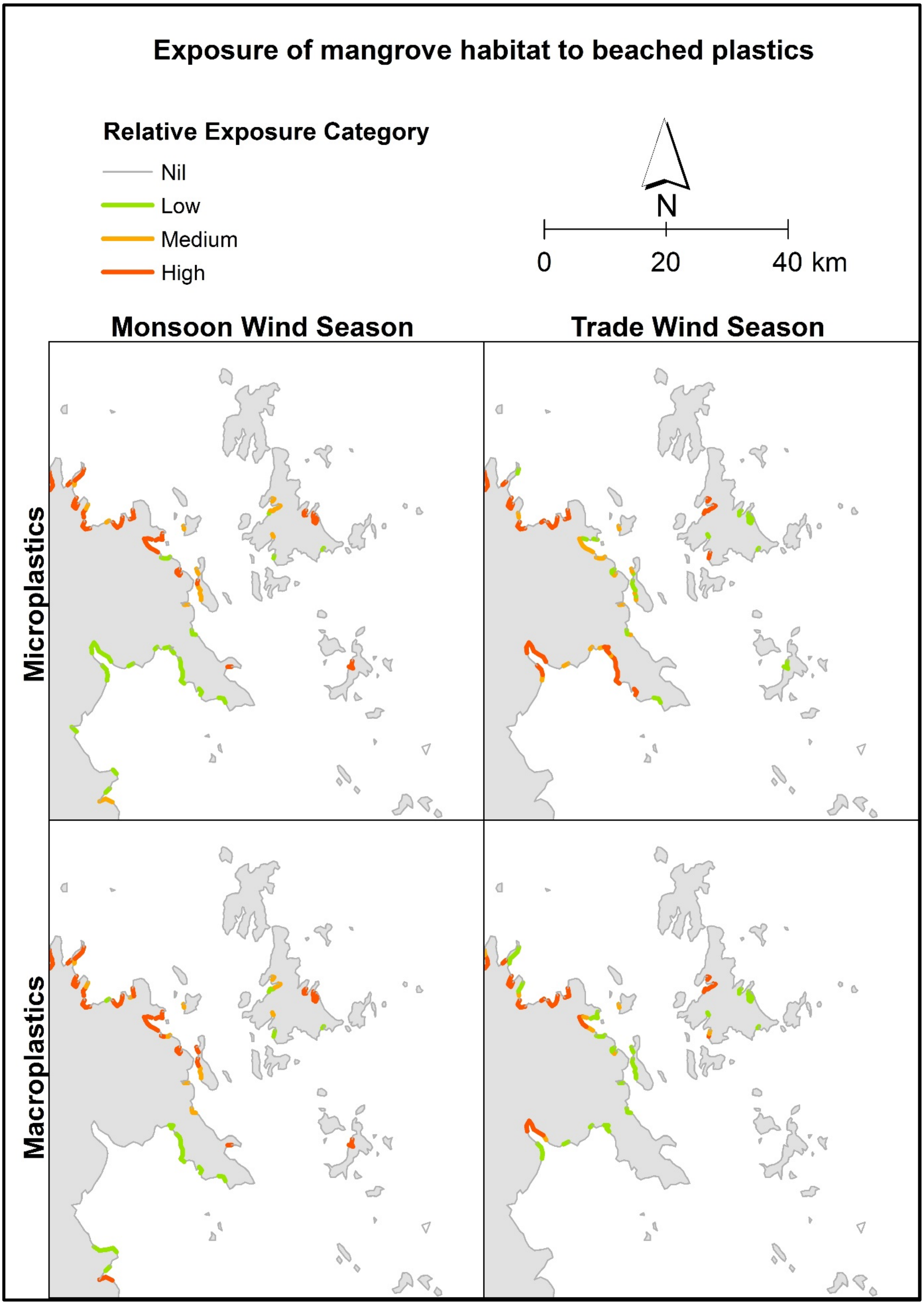

Figure 3.10: *The relative exposure to macro- and micro plastics for mangrove habitats in the Whitsunday region*

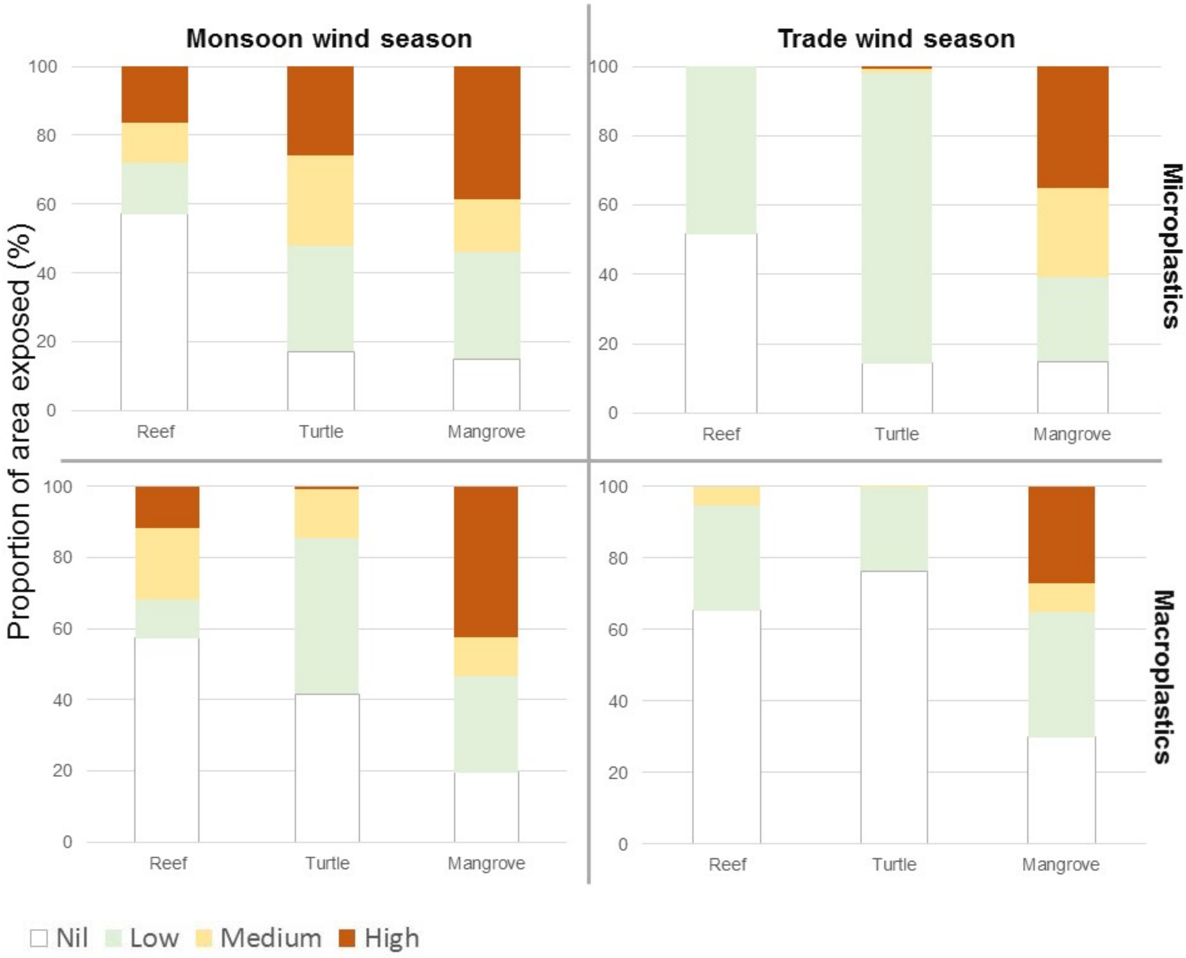

Figure 3.11: *The proportion of habitat area in each threat category by season and plastic type. The threat categories colours correspond to the exposure maps (Figure 3.6, 3.8, 3.10). The top panels show microplastics, and the bottom panels show macroplastics. The left column shows the Monsoon wind season and the right column shows the trade wind season.*

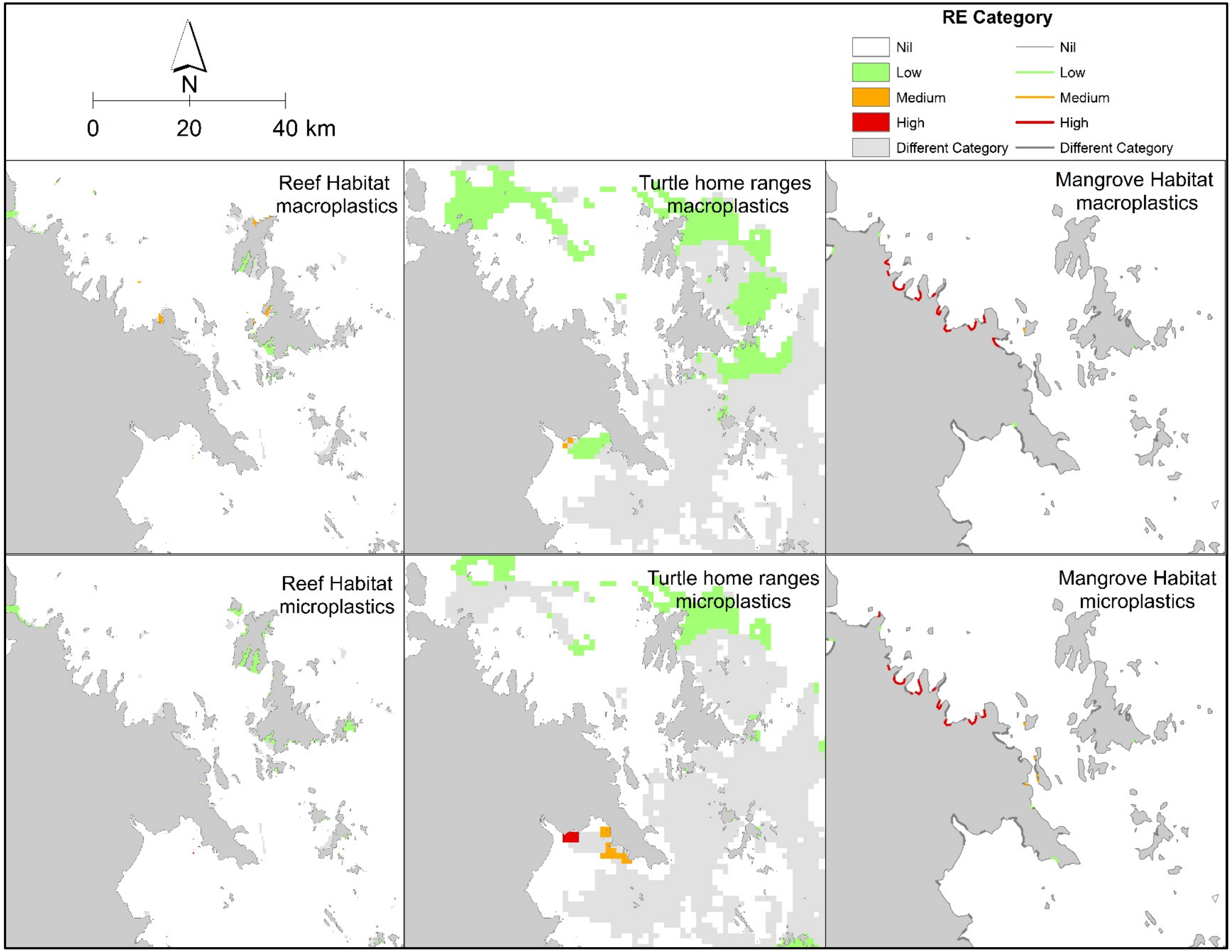

Figure 3.12*: Areas in each habitat that are in a consistent exposure category across wind seasons.*

3.4 Discussion

In this chapter I present a method of predicting the potential exposure of coral reef systems, mangrove habitats, and foraging flatback sea turtles to plastic pollution in a complex, coastal environment. I found that accumulation and exposure levels differed during each wind season (trade and Monsoon) and plastic type (macro- and micro-). The Monsoon wind season (October - March) resulted in the largest area of coral reefs, mangroves and turtles in the highest exposure category. In the trade wind season (April - September), plastics are pushed by the south-easterly wind out of the study area, reducing the relative exposure compared to the Monsoon wind season where the plastics are not pushed from the study area.

During the trade wind season, the plastics are moved out of the study area by local water circulation patterns, reducing the potential interaction between plastics and coral reefs, mangroves and turtles in the region. However, it is possible that plastics from external sources are imported into the study area during the trade wind season, which the current model did not capture. During the trade winds a more southerly wind direction would move any buoyant objects from the south or south-east into the study area (Critchell et al., 2015). In the Monsoon wind season, the wind-driven currents move plastics into areas that are protected from the typically strong trade winds. Fringing reef habitats are often found on the lee side of islands (Hopley, 1982; Kennedy and Woodroffe, 2002), and pervasive exposure to wind-generated waves can damage the coral structure and restrict growth. I found that, during the Monsoon wind season, sheltered reef habitats were more exposed to plastics, being moved by the, often calmer, Monsoon winds.

A large proportion of mangrove habitat had a relatively high exposure to plastic pollution in both wind seasons and for both plastic types. However, the risk posed by macro- and microplastics is likely to be different. For example, a single large object can damage a comparatively large area of mangrove habitat, for example, plastic sheeting or fishing gear (Goldberg, 1997; Uneputty and Evans, 1997). Microplastics significantly affect the sediment permeability in beach sediments at a concentration of approximately 16% plastic by weight (Carson et al., 2011), the concentration of microplastics (i.e. the number of plastics per area of coastline) required to make an impact on mangroves could be larger than it is for macroplastics. Even though a similar area of mangrove habitat is in the high exposure category for both macro- and microplastics, the threat posed by those categories (i.e. the consequence) may be vastly different.

In each wind season, a larger area of coral reefs were exposed to the highest exposure category of microplastic rather than macroplastic pollution. Microplastics affect reef habitats as they can be ingested by reef animals, including reef-building corals, for example *Dipsastrea pallida* (Hall et al., 2015). However, it may be that macroplastics have a larger and more immediate consequence to reef habitats. For example, scouring and smothering by large plastic net items impact the reef structure by damaging corals on a larger scale than individual coral polyps or colonies. Chiappone et al., (2005) found up to 11 individual reef organisms damaged by a single piece of derelict fishing gear in the Florida Keys. Lamb et al., (2018) found that macroplastics also increase the prevalence of disease in reef-building corals in several reefs across the Indo-Pacific region. It is important to note, that the types of debris described in these studies are often fishing gear that has been deposited *in situ* and therefore unlikely to be transported in the same way as the simulated debris in my modelling. Therefore, it is important for future studies to consider the types of plastics impacting sensitive habitats, and model debris with appropriate parameters to capture the true exposure.

Coral reef habitats in the Whitsunday region already face several pressures, and were recently exposed to moderate levels of bleaching coupled with cyclone damage from severe tropical cyclone Debbie in 2017 (Hughes et al., 2015 and unpublished monitoring data - http://www.abc.net.au/news/2017-07-13/comparison-photos-striking-damage-great-barrier-reef-cyclone-qld/8702192). Hence, it is clear from my results that macroplastic pollution adds an additional threat to reef habitats in the region and pervasive exposure to plastics could affect natural recovery after environmental disturbance.

Like coral reefs, flatback turtle habitat had a larger area exposed to the highest exposure category of microplastic than macroplastic pollution in both wind seasons. The degree to which plastic particles are ingested by fauna, such as flatback turtles, depends on the relationship between plastic size, abundance and gape size of the turtles. However, because microplastic particles in the environment vary greatly in size, shape and colour they pose a potential issue for all size classes of marine turtles (Schuyler et al., 2012; 2014). In general, the smaller the plastic particle in relation to the size of the digestive tract, the greater the potential that the particle will be passed through the gastrointestinal tract and cause less physical damage. In comparison, larger particles are more likely to cause issues during digestion, leading to physical injuries and sometimes death (Parga, 2012; Di Bello et al., 2013). While hard macroplastic items are not often found in the digestive tracts of turtles, turtles have been recorded biting pieces off softer macroplastic items such as polystyrene buoys/foam and plastic bags (Schuyler et al., 2012), the results of which can be similar to microplastic ingestion. This is important because both plastic bags and polystyrene buoys are commonly removed from beaches and storm water drains during clean-ups. Entanglement in derelict fishing net is commonly associated with turtle mortality in some areas of Australia (Wilcox et al., 2013). However, in the GBR rates of entanglement are low (Biddle and Limpus, 2011), and ingestion is a far larger, and largely unquantified issue.

It is clear from my analysis that, large areas of one of the most important foraging areas for flatback turtles in the Great Barrier Reef is exposed to high levels of plastic pollution during the monsoon season. Because flatback turtles are non-specific carnivores, foraging across the water level, and have differences in diet between age classes, exposure to microplastics is generally believed to increased rates of ingestion (Vegter et al. 2014; Wildermann, 2017). Plus, the inability of turtles to regurgitate mean that exploratory bites tend to be swallowed and thus plastics get ingested regularly (Schuyler et al., 2012). Hence, given the high exposure to microplastic in the monsoon season, the chances of encountering and ingesting plastic particles is likely to be high. Flatback turtles are one of six turtle species that live in the GBR region. Green turtles (*Chelonia mydas*), common throughout the GBRWHA, have very high inter-annual foraging site fidelity, especially around reef and mangrove habitat (Shimada et al., 2016). Conversely, flatback turtles are more mobile often moving between habitat types to forage (Wildermann 2017). Thus, assessments of risk at the scale conducted in this chapter are realistic. Also, while green turtles are largely herbivorous, flatback turtles are more opportunistic carnivores foraging across the water column – hence, looking at their exposure to suspended plastics is realistic. The exposure approach presented here is important because, while it does not indicate which individuals are most at risk, it enables a region-wide understanding of the degree to which an important flatback turtle foraging habitat may be exposed to plastic pollution, and in which times of the year. Ingestion of plastics is listed as one of the key threats to marine turtles in Australia, yet little is known about the degree to which different species and important habitats are exposed (Commonwealth of Australia 2017). This chapter represents the first time the degree to which an important foraging habitat for an Australia sea turtle species is exposed to microplastic pollution has been quantified.

Despite turtles often being used as flagship species to highlight the problems of plastic pollution, there is limited information about their exposure to plastic pollution. A global risk assessment by Schuyler et al., (2016) assessed the hotspots of plastic ingestion by marine turtles, however, the broad spatial scale used (1^0 x 1^0) is unsatisfactory for local governance. Wilcox et al., (2013) used known ghost net sink locations to predict distributions of ghost nets in the Gulf of Carpentaria, Australia, combined with estimated turtle abundance and known turtle entanglement rates, to model the risk of turtles to ghost fishing. The spatial scale used for the risk analysis was at a resolution of 5^0 latitude by 5^0 longitude, which, while useful for their study, was much broader than the resolution I used. A coarse spatial resolution is useful across a large geographic range, such as to understand the issue of ghost net entanglement in the Gulf of Carpentaria. However, it would be insufficient for addressing local, small-scale management questions relative to the Whitsunday region. In this chapter I use a resolution of 1 km^2 and 330 km^2 to undertake the analysis meaning that the results could be used to aid decisions made by agencies managing the Whitsunday area. One such management action could be targeting debris removal activities on specific beaches. One of the main contributors of microplastics in the environment is the physical breakdown of macroplastic items on beaches. The microplastic particles can either remain mixed with the sand or resuspended back into the aquatic habitats. Removing macroplastic items from the Whitsunday and other GBR beaches, while not likely to be a cost-effective long-term solution, could reduce local inputs of microplastics, and reduce the existing load in local waters. Furthermore, given the locations of most of the turtle nesting beaches in Australia are well known, the modelling approach I used could be developed to conduct a risk assessment related to the exposure of nesting turtles on beaches with high macro- or microplastic loads, as has been done for examining exposure to light pollution (Kamrowski et al., 2012). The method presented in this chapter could be easily modified for this purpose.

I acknowledge that the approach presented in this Chapter has limitations that will prevent it being useful to management immediately. For example, the modelling underlying the exposure analysis has not at this stage been ground-truthed to field data, and sources have not been quantified. As a result, the predictions could be inaccurate in magnitude and in the spatial patterns, however, the SLIM system was used to predict accumulation areas along the whole Queensland coast (Critchell, et al., 2015) and approximately represented the available field data. Even with these limitations, this chapter presents a novel approach to exposure analysis for plastic pollution on a small, management-relevant scale. This approach is readily scalable to larger jurisdictions and other habitats or taxa. In this study I have not presented consequence information for each habitat, and how consequence is likely to change at increasing exposure levels. Without these data, actual risk cannot be presented, only potential risk.

There is a strong need to quantify the consequence of each level of exposure to understand the true risk that plastic pollution poses to habitats and species because the interactions between animals and a single microplastic or the animal and a single macroplastic are not equal. Plus, in relation to coral, some species may be more susceptible to impacts than others, e.g. branching species and fishing line entanglement, and for turtles, smaller age classes may be more susceptible than adults. Another consideration for the assessment of exposure to habitats compared to species (i.e. marine turtles) is that mangroves and reefs remain in the same geographic location and thus plastic impacts can accumulate. However, turtles are mobile and will use different habitats daily. Turtle recovery plans and the regional management units (RMU) need to be informed at smaller scales and higher resolution (Wallace et al., 2011). Relating the exposure categories presented here to the consequence of that exposure to the habitat or population, is a vital next step in understanding the risk of plastic pollution.

In this chapter I presented an exposure analysis approach that could be used for a variety of habitats and species known to be impacted by plastic pollution. I found that, in the Monsoon wind season the habitats and species were highly exposed to plastic pollution, with few consistencies with the trade wind season. The exception was for mangrove habitats which had areas of consistently high exposure across wind seasons. The implications are important, especially for flatback turtles for which marine plastic ingestion is a recognised threat, and for coral reefs in the region which are already stressed from coral bleaching and cyclone impacts. Once validated with field data, the exposure maps presented can be used in the prioritisation of conservation resources, from debris removal programs to locating offset initiatives. The framework I used could also be refined to map the exposure of a particular plastic class, or types of objects that are known to impact a particular species or habitat e.g. drifting net on reef structures.

Chapter 4
Using field data to interrogate a plastics dispersal model

Plastic pollution is suffocating the ocean and coastal environments. Understanding the accumulation of plastic pollution in the coastal zone will improve targeted management action. Quantifying inputs of plastics from various sources at local scales is imperative to successful modelling of plastic dispersal. In Chapter 2, I presented a novel hydrodynamic model for plastic movement in the ocean. In Chapter 3, I used that model to conduct an exposure assessment of plastic pollution for coral reefs, mangroves habitats and flatback sea turtles. The assessment raised questions about the consequence of plastic exposure (see Chapter 5), and the source and input parameters used in the model. In this chapter, I build on the hydrodynamic model developed and presented in Chapter 2 in order to identify the likely sources of plastics at a management-relevant scale, and to explore the environmental conditions and processes that affect plastic accumulation on a complex coastline. To do so, I used field data for macro- and microplastic particles to interrogate the model. I found marine sources are likely to be more important than coastal sources for microplastics, and coastal sources are more important for macroplastic than microplastic particles.

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4.1 Introduction

To mitigate the impacts of a dispersive and persistent pollutant like plastics it is imperative to understand the sources and accumulation areas (sinks) of the pollutant (Kako et al., 2011; Carson et al., 2013). Dispersal and accumulation models are useful for understanding the way pollutants will move through and impact the environment, and they are integral to the development of effective management actions. For example, modelling has been used in risk/threat analysis by predicting accumulation of various types of plastic pollution and comparing this with areas of interaction with vulnerable species (e.g. Chapter 3; Wilcox et al., 2013; Halpern et al., 2015; Darmon et al., 2017). Sherman and van Sebille (2016) have also used models to predict the accumulation of microplastics to identify maximum efficiency placement of plastic removal devices. However, managing the inputs of plastic pollution is generally a responsibility taken at municipal levels, which requires local-, rather than global-scale models. Currently, local models are lacking because of challenges incorporating the multiple factors influencing the dispersal pathways and accumulation of plastics in the coastal environment, including the supply of plastics to the area, the rate of plastic loss from the area, and the processes that move plastics though the area. Each of these factors represent key knowledge gaps in plastic dispersal and accumulation.

In addition to the complexities of hydrodynamic processes in coastal regions, different objects can have different dominant driving processes moving them through or across the water column. For example, the movement of larger objects floating on the ocean surface will be strongly affected by wind whereas objects below the water surface will be most strongly affected by currents (Breivik, et al., 2011; Fazey and Ryan, 2016). Objects on beaches, or on the substratum sub-tidally, are additionally affected by substratum properties (Carson et al., 2011). Understanding these differences is important when considering and comparing the movement of macroplastics and microplastics, which could have the same density but the differences in their size and shape would alter how the plastics move in an aquatic system (Daniel et al., 2002; Isobe et al., 2014). This may result in different sinks for macro- and microplastics even when they originate from the same source, but these dynamics are poorly understood. In one study, Kako et al., (2010) demonstrated that incorporation of field data could result in estimation of the source locations of specific objects. However, that study focussed on just one type of macroplastic (disposable cigarette lighters) and the results are difficult to generalise to other types of macroplastic or microplastic.

Combining hydrodynamic models with field-based collection and observations at specific locations is a tractable approach to identify sources and sinks of plastics in the marine environment. The alternative approach of identifying the source of plastic items collected from the environment (at sink locations, e.g. Tudor and Williams, 2004; Topçu et al., 2013) is time consuming and can only be applied to a subset of items found at the sink location. Direct identification of plastic sources is possible in local river systems by collecting samples in the river channel and estuary environment (e.g. Moore et al., 2011). Conversely, quantifying the input from sources a long distance away from the reference location (external to the local system) is extremely challenging due to the persistence of plastics in the environment, and the ability of plastics to be moved great distances. For some plastic items, barcodes or foreign labels (e.g. Topçu et al., 2013; Duhec et al., 2015; Smith et al., 2018) might be able to identify the manufacturer or the country of sale for the product, but rarely can they identify the location of its use and disposal. Another complication is that the number of objects with labels or barcodes remaining intact and readable at a clean-up site is often low (as seen in Duhec et al., 2015; Smith et al., 2018). These challenges could be especially prominent in areas with high volume international shipping, or areas adjacent to large urban developments. Using hydrodynamic models can overcome these limitations because they mathematically predict the movements of the ocean, from which we can predict the movement of submerged and floating objects and thus identify source and sink areas (Lebreton et al., 2012). However, using modelling alone, in the absence of field data, is problematic because model-derived predictions of dispersal or accumulation can be sensitive to the parameters used in the model (Chapter 2), many of which cannot be easily measured in natural systems.

There are two ways to understand sources through using a hydrodynamic modelling approach: 1) hindcast modelling with reverse dispersal (e.g. Isobe et al., 2009; Reisser et al., 2013); or 2) forecast modelling with an advection-dispersion approach (Lebreton et al., 2012). There are advantages and disadvantages to both approaches. With hindcast modelling it is possible to predict the source candidates of a known sink location through backward-in-time models (Reisser et al., 2013). However, the further back in time the model is run, the larger the prediction error becomes, and so the confidence that the source is correctly identified is reduced. The "gold standard" method is to run subsequent forward models to narrow down the source candidates, and enable identification of a more precise source (e.g. Isobe et al., 2009; Kako et al., 2011) but this process has not been achieved at a resolution adequate for local management action, nor has it been applied to microplastic pollution (Hardesty et al., 2017). It is also the case that some hydrodynamic models do not have the capacity to hindcast, as the mathematics are complex. Application of hindcast modelling also requires knowing the arrival time of the objects at the sink location (Griffin et al., 2016), information that is rarely available for marine plastic debris. Forward modelling is the alternative approach wherein, if the possible source locations are known, a particle tracking scheme (e.g. Lagrangian) can be used to predict the sinks for plastics from those source locations (e.g. Griffin et al., 2016). This method is advantageous because the user can accurately parameterise horizontal mixing of the water due to turbulence (essentially a random process) (Andutta et al., 2013; Mao and Ridd, 2015). However, the disadvantage of the forecasting approach is that it requires simulation of plastic dispersal from all potential sources, and simulation under the environmental conditions specifically relevant to the likely arrival times of the plastic debris. As weather conditions have a large influence on the movement and accumulation of plastic pollution (Storrier et al., 2007), different plastic accumulation patterns are frequently observed in different seasons (as seen in Chapter 3), and among different locations depending on exposure to prevailing wind and swell (Storrier et al., 2007; Critchell et al., 2015).

Plastics found in the marine environment are generally from either local land-based or marine sources. Land-based sources include urban rivers and storm drains (Castañeda et al., 2014; Eerkes-Medrano et al., 2015; Cable et al., 2017), or beaches (Claereboudt, 2004), while marine sources include shipping or fishing actives (Chen and Liu, 2013; Bilkovic et al., 2014), or simply the 'stock' of plastic now present in our global oceans (Dameron et al., 2007; Carson et al., 2013). Understanding the dominant source supplying plastic debris to a location can inform management decisions because the knowledge can aid priority setting, such as actions to reduce local inputs if local sources are dominant, or by cleaning beaches to maintain ecosystem services if external sources are dominant, while initiating collaborations with external agencies to reduce inputs at the source. However, identification of sources and sinks of plastics is complicated by the physical geography of the local area, and the influence this has on the processes driving the accumulation at specific sites. For instance, sites with complex topography are likely to have higher variability in accumulation, and in plastic retention, due to small-scale water movements (Daigle et al., 2014) and the blocking influence of coastal features. Therefore, it may be more difficult to model the accumulation of plastics at a complex site than at sites with relatively simple hydrodynamic and coastal features (Ballent et al., 2013). The surrounding physical geographic features can also influence the supply of plastics to a site. For example, it is possible that blocking features, such as islands, can create a "supply shadow" in the downstream locations. Finally, if plastics have become ubiquitous in the marine environment it is also possible that using only point-source locations are no longer appropriate for understanding plastic accumulation in local areas. Resolving the relative effects of source location, local topography, and seasonal effects requires hydrodynamic simulation experiments exploring each of these factors.

In this chapter I will use field data to interrogate the SLIM plastics hydrodynamic model (Chapter 2) to deduce the most likely source (local or external) of the plastic accumulating on beaches in the Whitsunday region (Chapter 1, Section 4), and to better understand the processes influencing plastic accumulation. I also aim to enhance our ability to accurately represent this accumulation with a fine-scale hydrodynamic model. I will do this by comparing field-collected micro- and macroplastic data to predictions of plastic accumulation on beaches given dispersal from specific source locations, dispersal from a diffuse source, and dispersal under specific weather conditions. I treated macro- and microdebris separately in both the model predictions and field data because the processes that drive dispersal and accumulation of these types of debris are different (Isobe et al., 2014). The results generated in this chapter will improve our understanding of the physical processes that dictate plastic accumulation in the coastal zone, advance our use of hydrodynamic models to predict accumulation hotspots, and improve our ability to provide an empirical basis for management action.

4.2 Methods

4.2.2 Overview of approach

To achieve the aims of this chapter required robust field data quantifying plastic accumulation at specific locations within the Whitsundays region. For macroplastics, I used an existing dataset collected by Eco Barge Clean Seas Inc. (see section 4.2.5 below). For microplastics, I collected and analysed beach sediment samples from 18 locations around the Whitsunday's region during August 2016 (see section 4.2.4 below). To select sites for microplastic sampling, I first used a hydrodynamic model to simulate dispersal and generate predictions of the spatial distribution of microplastics. I then used the model's outputs to identify potential areas of high microplastic accumulation (hotspots) versus low accumulation (coldspots), and I subsequently collected sediment samples from those locations. To identify the most likely sources of these plastic types, I then compared model predictions of macroplastic and microplastic accumulation from land-based versus marine sources with the field data, with the assumption that the model predictions that provided a better fit with the field data would be generated from the more likely source. These comparisons were made in two phases. First, I tested whether the model correctly predicted the location of hotspots and coldspots, and second I tested the correlation between the measured and predicted numbers of plastics which accumulated at each location. To understand the processes driving accumulation, I compared the field data to predicted accumulation from model scenarios using different weather conditions and, additionally, under dispersal from a diffuse source. This final source scenario was included in order to determine if using point sources was effective for modelling local plastic pollution distribution, and to assess the effect of site exposure on plastic accumulation.

4.2.3 Modelling scenarios

I used the version of the SLIM model presented in Chapter 2, which incorporates the physical processes affecting plastic dispersal, including beaching, re-floating from the coast, settling to the bottom, as well as degrading from macro- to microplastics. These parameters may be important since each influences the transport and accumulation of plastic particles in the marine environment (Zhang, 2017; Chapter 2). In the model, simulated plastics have two states: 1) macroplastics which are assumed to be moved by the vector sum of the wind and currents; and 2) microplastics which are assumed to drift as neutrally buoyant objects, moving with the currents only.

The field samples I collected for microplastics represent the net microplastic accumulation (input minus losses) over an unknown time period, and therefore over unknown environmental conditions. To assess how variation in environmental conditions potentially affects microplastic accumulation in the time leading up to the field sampling, I generated four sets of plastic (macro- and micro-) accumulation predictions using forcing climate data for four different 45 day time periods, in February, March, June and August (Table 4.1). Forcing climate data included wind speed and direction and sea surface elevation input data (as per Chapter 2). As the sampling events for macroplastic quantification occurred sporadically (see section 4.2.5 below), the use of these time periods enabled me to compare the field data with a set of typical conditions experienced in the region.

I used the same model inputs and parameters as the model described in Chapter 3, where simulated plastics were released from locations that were the most likely to be sources of plastic pollution in the Whitsunday Region, e.g. river mouths, tourist beaches and shipping lanes (Chapter 3; Figure 4.1). To tease apart the importance of the dominant plastic sources of the Whitsunday region, the particles originating from each point source were pooled into land-based (local) sources and marine (external) sources. I note that particles from these two broad sources are either within (local sources) or outside (external sources) the control of the local municipality.

As shown in Chapter 2, model predictions are strongly influenced by source location and, therefore, the role of other model processes can only be determined if the ‘source effect’ is removed. To assess the influence of exposure of sites to prevailing winds and waves on plastic accumulation, I ran an additional scenario with a uniform source (the grid scenario, Table 4.1), where particles originated at each vertex of a 10 x 10 km grid and predictions of accumulation at each site were pooled across the entire grid to give one site-specific accumulation value per simulation day.

As described in Chapter 3, I set some parameters in the model to constant values. I imposed a constant wind shadow in the lee of the islands of 2500 m because implementing a variable wind shadow would require coupling a wind field model to the SLIM model. In reality, the size of the wind shadow would change with the size and shape of the land mass causing it (Myksvoll et al., 2012). However, in the context of my study a constant wind shadow was acceptable because model predictions are not strongly affected by the length of the wind shadow (parameter ranked 8.75 out of 15 see Chapter 2). The model was also forced with a standard M2 tide inflow and forcing from the Coral Sea; both are idealised but have been successfully used in previous studies to provide an acceptable representation of water movements (Hamann et al., 2011; Andutta et al., 2013; Critchell et al., 2015). The scenarios were run for 45 days, this allowed the particles to be well mixed while remaining within the study area, as the majority of particles have left the domain after this time. The parameter estimates used in each of the scenarios are presented in Table 4.1.

The model generated outputs for each of six particle categories, three each for macro- and microplastics. The three categories were: 1) beached particles, 2) suspended particles, and 3) settled particles. The modelled results for beached particles are the most appropriate to compare with the field data collected from beaches so only the model outputs for the beached particles are considered in this chapter. For each scenario the model gives a daily value of particles throughout the entire study area. I extracted the particles present at each site using an automated script of my design which counts the particles of all types within a user-defined box that represents the site for each day of the simulations (Appendix 1). The number of particles of each type present at each site was then transformed for comparison to the field data (described below for simulated macro- and microplastics).

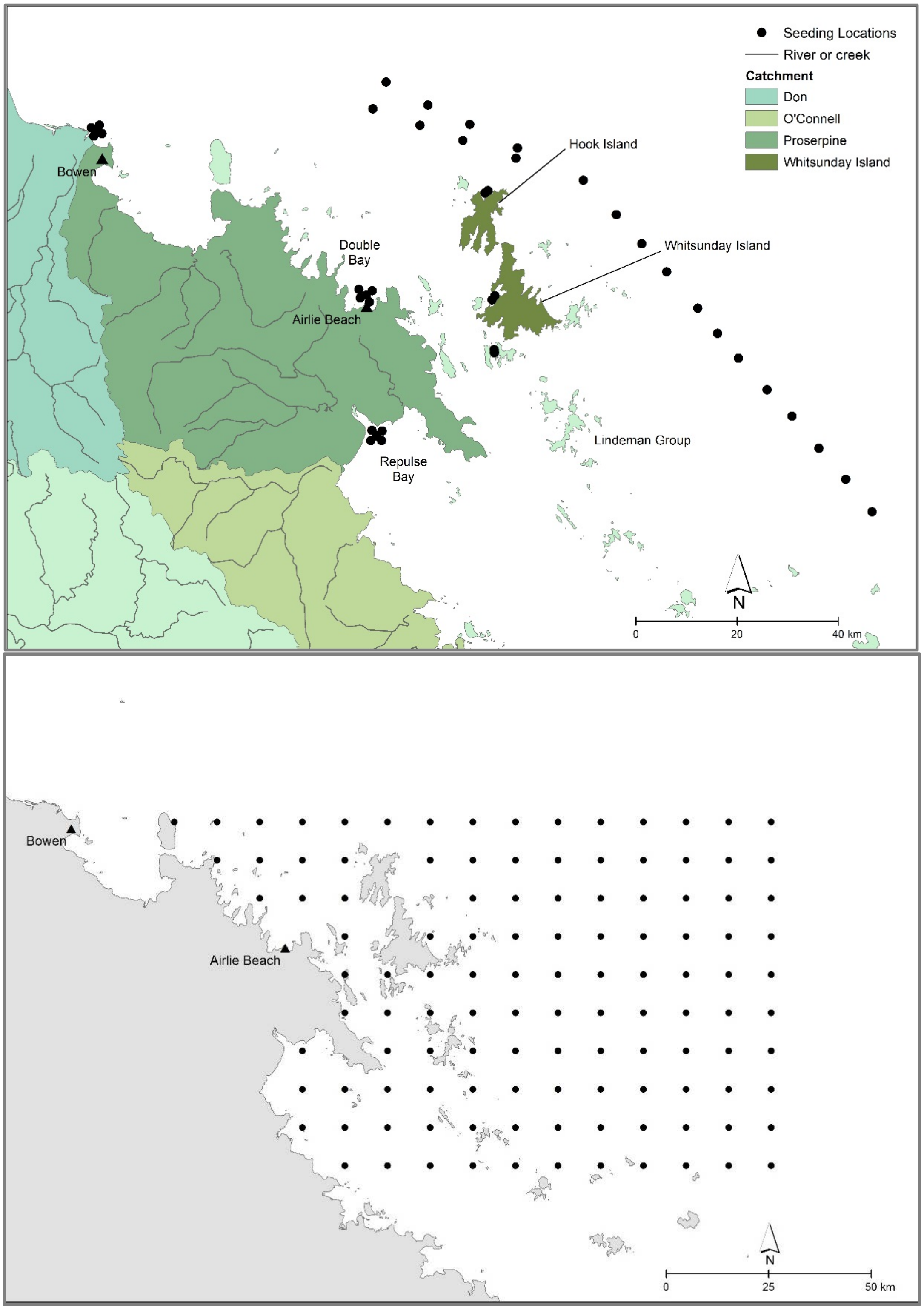


Figure 4.1: *Map showing the study area. Top panel shows the seeding locations used in the scenarios (black circles), the rivers (grey lines) and the catchment (green hues). The bottom panel shows the seeding locations used for the grid scenario.*

***Table 4.7**: table of scenarios of modelled plastic movements. Columns show the values of each parameter used in each scenario and sources for these parameter estimates are described in Chapter 2. * shows the scenarios used in prioritising field sites.*

Scenario Name	**source location**	**Forcing data start date**	**Length (days)**	**Wind shadow length (m)**	**Number of seeding locations**	**Particles per seeding location**	**Degradation on land (% per day)**	**Degradation at sea (% per day)**	**Resuspension probability (% per day)**	**Justification**
February*	Land	Feb-2014	45	2500	21	10,000	0.00001	0.000001	0.2	First simulation used to choose field sites (Figure 4.1A)
	Marine	Feb-2014	45	2500	20	10,000	0.00001	0.000001	0.2	First simulation used to choose field sites (Figure 4.1A)
March*	Land	Mar-2016	45	2500	21	10,000	0.00001	0.000001	0.2	Forcing data 5 months prior the microplastics field campaign
	Marine	Mar-2016	45	2500	20	10,000	0.00001	0.000001	0.2	Forcing data 5 months prior the microplastics field campaign
June*	Land	Jun-2016	45	2500	21	10,000	0.00001	0.000001	0.2	Forcing data 2 months prior to the microplastics field campaign
	Marine	Jun-2016	45	2500	20	10,000	0.00001	0.000001	0.2	Forcing data 2 months prior to the microplastics field sampling
August	Land	Jul- 2016	45	2500	21	10,000	0.00001	0.000001	0.2	Forcing data before and during the microplastics field sampling
	Marine	Jul- 2016	45	2500	20	10,000	0.00001	0.000001	0.2	Forcing data before and during the microplastics field sampling
Grid	Uniform grid macros	Jul- 2016	45	2500	117	10,000	NA	NA	0.2	Uniform distribution to better understand the driving processes for macroplastics (Figure 4.1B)
Grid	Uniform grid micros	Jul- 2016	45	2500	117	10,000	NA	NA	0.2	Uniform distribution to better understand the driving processes for microplastics (Figure 4.1B)

4.2.4 Microplastics field data

Site selection

The selection of field sites was based on the combination of three modelled scenarios incorporating a variety of weather conditions (denoted by * in Table 4.1). Using the output of the last day of the modelled scenarios, the locations of the simulated beached plastics were used to identify beaches most likely to accumulate plastic particles (hotspots) and the most likely to be clean (coldspots). To assess the model predictions of accumulation, all beaches in the study area were categorised into nil, very low, low, medium and high accumulation categories based on the spread of data for the individual scenario, using the "natural jenks" function in ArcGIS (ESRI, 10.2). Hotspots were chosen as locations with medium to high predicted accumulation in at least one of the three scenarios. Coldspots were chosen as locations with nil to very low (deemed insignificant) predicted accumulation in all scenarios, and were in relatively close proximity to the hotspot sites to minimise travel time between sites during field work (see Figure 4.2). Originally, I chose 20 sites for field data collection. Two sites, one hotspot and one coldspot, had to be abandoned due to weather and logistical difficulties.

Sample collection techniques

At every beach site I collected three replicate samples from three locations (nine samples per beach) along the highest high tide mark. The sampling locations were 100 meters apart or approximately equidistant for smaller beaches. The replicate samples were taken within 1 m square at each of the three locations. Depending on beach type, the sediment samples were either collected in a 25x25x2 cm quadrat (~0.00125 m^3), or a core sample of 8 cm in depth and diameter (~0.0004 m^3). The priority was given to the core samples because they provided a more consistent volume of sediment whereas, using quadrats, I found it difficult to collect a consistent depth (and therefore volume) of sediment. However, if the bedrock was close to the surface and the core was impossible (3 out of 18 sites), three replicate quadrats were taken at each sample location along the highest high tide mark (total samples N = 162).

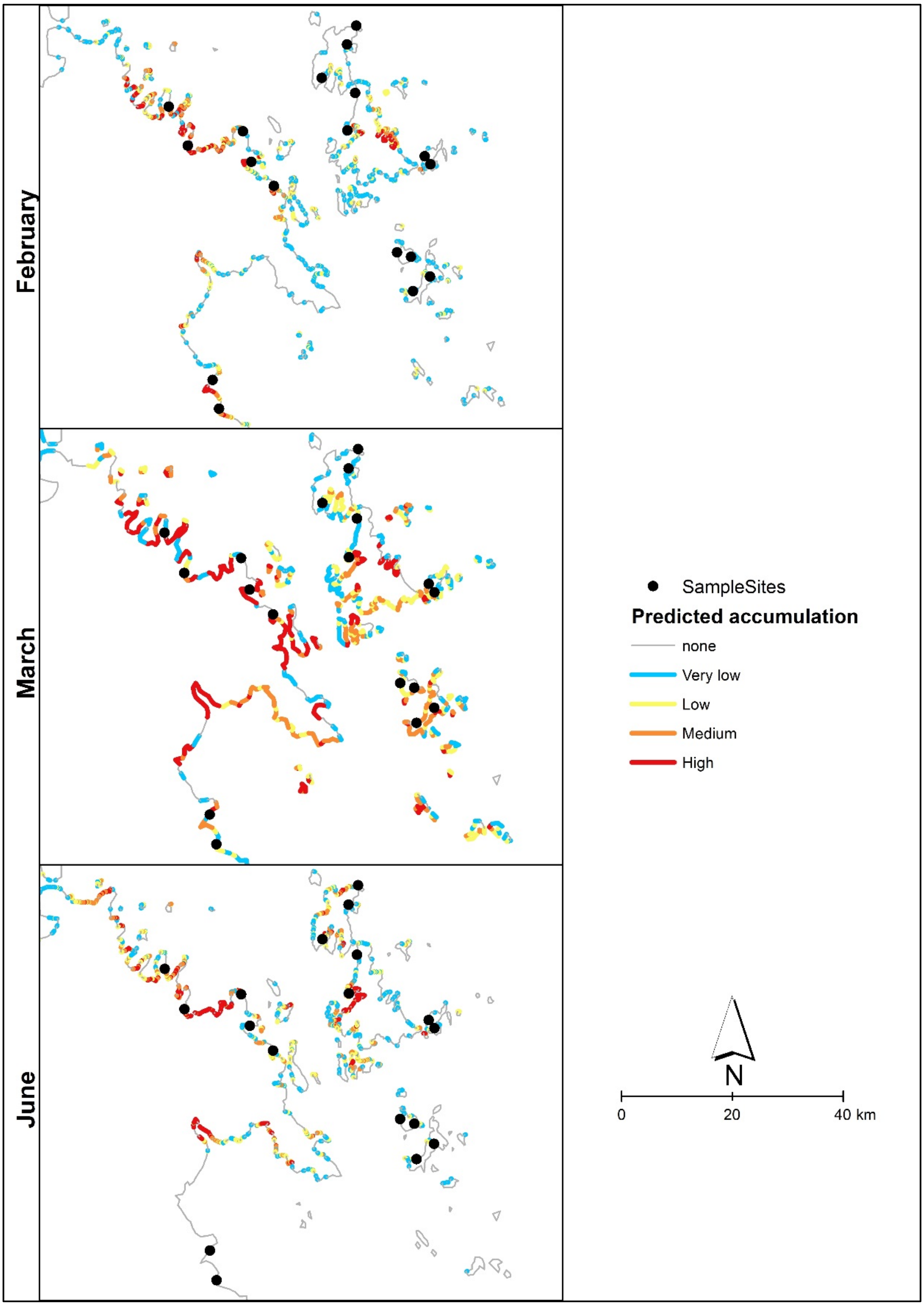


Figure 4.2: *Map showing the predicted accumulation for each scenario used in the field site selection. Sites selected for field sampling (black circles) were based on these three sets of simulations (Feb, March, and June). Hotspots had "medium" to "high" predicted accumulation in at least one set of simulations, and coldspots had zero to "very low" predicted accumulation in all three sets of simulations.*

Processing sediment samples

To quantify the microplastics in the sediment samples, I processed the sediment using a method modified from Claessens et al., (2013). The samples were first run though the elutriation column (or fluidised sand bath) for 15 minutes using a flow rate of approximately 300 L $hour^{-1}$ (Claessens et al., 2013). The lowest density particles from within the sample were collected in a 36 µm sieve under the outflow of the elutriation column. These were then transferred to a beaker via washing of the sieve with salt water. The low density particles from the elutriation column, containing the vast majority of the plastics from the sample (Claessens et al., 2013), was further separated by suspending the sample in concentrated potassium and sodium chloride solution (density at least 1.8 g cm^{-3}) and leaving it to settle. The buoyant particles were siphoned off and filtered onto glass-fibre filter papers. This high-density fluid separation process was repeated three times to ensure maximum plastic removal from samples. Depending on sediment sample type, the number of filter papers varied (from four to 18 used for one sample). Samples that had high silt or mud content required a smaller volume of the suspension to be filtered on to each filter paper to ensure accurate plastic identification (i.e., the sample required more filter papers). At two of the sites (Saba Bay and Hazelwood Island - South East site), the highest high tide line was dominated by pumice stone. The density separation technique was not appropriate for these samples because pumice stone is less dense than plastics. For these samples, manual sorting was the only viable option. I sieved the samples into >5 mm, 5-2 mm, and 2-1 mm portions, the larger size classes were sorted by eye and the smallest size class was sorted under the dissecting microscope. The size class <1 mm was processed using the high density fluid stage described above.

I visually inspected the filter papers using a dissecting microscope set to 1.5 x magnification, only increasing the magnification to clarify the categorisation of a particle. I used this method to limit the minimum size of particles that I was categorising (~250 µm). All plastics were identified by eye in a conservative manner: if there was any doubt as to the particles material nature it was not counted as plastic, this likely resulted in an under-estimation of the microplastic abundance. Results, therefore, are likely to underestimate the total number of microplastics present at each site but still enable comparison of the relative abundances of microplastics among sites. All samples were retained for analysis by Fourier transform infrared spectroscopy at a later stage to confirm their identity. I sub-sampled the counting and counted the odd numbered filter papers, making at least half of the filter papers of each sample, as a subsample e.g. 9 out of 18 (papers numbered 1,3,5, etc.). Plastic counts per sample was calculated to provide an estimate of total microplastic abundance per cm^3 of sediment on each beach. From these data, the observed accumulation category of each site was created. Observed microplastic 'hotspots' were defined as sites with above the median observed accumulation and observed microplastic 'coldspots' were defined as sites with observed accumulation below the median value.

4.2.5 Macroplastics field data

I obtained macroplastic data from the Australian Marine Debris Initiative (AMDI) database (https://www.tangaroablue.org/database.html). The AMDI database stores counts of debris items in categories based on the material that the debris item is made of and its use. I accessed data for the Whitsundays region which was primarily collected and entered by a local NGO, EcoBarge using the AMDI method (https://www.tangaroablue.org/resources/how-to-manual.html). The raw data consisted of many types of marine debris, including wood and metal items that would disperse differently in the marine environment compared to plastic pollution. To ensure the data were suitable for comparison to the model outputs, I extracted object categories that are likely to behave similarly to the simulated plastics in my model. My chosen AMDI categories were: "Bleach and cleaner bottles", "bleach bottle (KKK type)", "lids & tops, pump spray, flow restrictor and similar", "pens markers and other plastic stationary", "plastic bits and pieces, hard and similar", "plastics drinks bottles (water, juice, milk, soft drink)", "toothbrushes, brushes and combs, hair ties, etc." . I chose these categories, as they are regularly found on the beaches of the Whitsunday region and easily identified by volunteers.

The macroplastic data still had inconsistencies that confounded direct quantitative comparison to the modelling outputs. As clean-up trips by volunteers happen sporadically, based on weather and availability of boats and volunteers, there were very few clean-up trips conducted at times consistent with the weather conditions used in the models. The diverse range of weather patterns occurring before the clean-up events makes comparing these data to the fairly specific conditions represented in the model difficult. To overcome this limitation I used repeat visits to the same location to calculate the accumulation rate at each beach. The number of plastics collected for each visit was divided by the number of weeks lapsed between visits, with the data from the first visit disregarded as it was considered an "initial clean" of plastics that had accumulated over an unknown time period. Data collected at subsequent visits were taken to be the plastics accumulated since the previous visit and an average accumulation rate calculated for each site. In addition, the nature of this citizen science data meant only "hotspot" beaches were sampled, as typically volunteer groups will only go to beaches with a large amount of debris to increase debris removal per unit effort. Data from sites with fewer than three visits were not analysed, consequently only eight sites fit the criterion of this study and could be used to perform the comparison to the model outputs.

4.2.6 Comparison of model outputs to the field data

I used a two-step process to compare the field data to the model predictions. First, I assessed whether the model predictions under each scenario correctly categorised the site as a hotspot or a coldspot based on the field data. To compare the microplastic field data to the scenarios the sites were categorised based on the median predicted daily accumulation value, those above the median were classed as hotspots and those below were coldspots. This was to ensure each scenario had nine hot and coldspots for consistency between scenarios. For the macroplastic data, I compared the observed accumulation rate with the model predictions of the weekly accumulation rate, i.e. the mean number of simulated plastics accumulated per week of the 45-day simulation. As all field data were assumed to be from hotspot sites, I used a threshold value of 40 particles per week to classify sites in each scenario into predicted hotspots or coldspots. The threshold of 40 particles per week was chosen as a representative high value of accumulation as determined from the range of accumulation values predicted from all the simulations.

The second step was to compare the predicted and observed values of accumulation. The predicted accumulation in microplastic scenarios was the median daily accumulation value across the 45 day simulation and the observed microplastic accumulation was total (per beach) plastics per cm^3 from the field samples. To compare the macroplastic accumulation, I used the weekly observed accumulation rate and the predicted weekly accumulation rate calculated from the model outputs for each scenario. I conducted Pearson's correlation analysis to compare the fit of the observed and predicted values for each scenario, with the assumption that the better the fit the more likely the scenario was representative of nature.

4.2.7 Site specific processes

Exposure to the wind/wind driven waves is often cited as the main cause for accumulation of plastics (Storrier et al., 2007; Critchell et al., 2015), however, waves are not currently included in the SLIM system. To do so would require coupling with a wave model, which is outside the scope of this thesis due to the difficulties coupling the two models in the core code of SLIM. Exposure at each site is different based on orientation to the dominant wind direction, and the fetch. These site-specific processes, not included in the model, could influence accumulation of plastic debris on the surveyed beaches, and lead to a lack of agreement between model predictions and observed data. To understand whether exposed or sheltered sites were more reliably predicted to be hotspots or coldspots, prediction success was compared to the degree of wave exposure of each site (see 'Relative Exposure Index' below). To assess whether exposure influenced plastic accumulation, the observed accumulation was also compared to the exposure. The influence of bay shape and the orientation of the site were also qualitatively compared with the observed accumulation and predictability to understand the influence of these factors. The prediction success metric was calculated as the proportion of scenarios that correctly categorised a site as a hotspot or coldspot.

4.2.8 Relative Exposure Index

To understand the role of exposure in plastic accumulation, I calculated the Relative Exposure Index (REI) for each site and compared this metric to the observed accumulation of both macro- and microplastics. REI is a standard measure of the exposure a location has to wave energy, based on the directional frequency of the wind and the fetch distance. Exposed sites (i.e. those that have long fetch distance in the direction of the dominant wind) have a high REI score and sheltered sites (i.e. those with a value of zero fetch distance in the direction of the dominant wind) have a small REI value. I calculated REI using the Generic Relative Exposure Model (GREMO) (Pepper and Puotinen, 2009). The method is as follows: at each site, a set of 16 radiating lines were drawn out to 650 km, beyond this waves are considered unlimited by fetch. The radiating lines were clipped to the closest wave blocking obstacle, and this becomes the fetch distance along the radial direction. The wind speed and direction were used to calculate the directional wind frequency along each radial direction. The exposure is the sum of these values.

$$REI = \sum_{i=1}^{16}(V \times P_i \times F_i)$$

Equation 4.1

Where *i* is the *i*th compass heading, V is the mean wind velocity, P_i is the frequency that the wind blows from the *i*th direction (%) and F_i is the length of the fetch line after clipping to the closest obstacle. For this analysis I used all available wind data (Jan 2010 to Sept 2016) which encompasses the modelling scenarios used here but also captures the average wind speed and direction of the region.

4.3 Results

4.3.2 Microplastics Field data overview

I found microplastics at every field site (Figure 4.3). The maximum number found in one sediment sample from Saba Bay was 907 plastic particles in 509 cm^3 of sediment. The smallest number of microplastics found was at Cape Conway where I found one plastic particle in one of the nine sediment samples. Consequently, the range in average microplastic density across sites varied from 0.002 to 1.78 plastics per cm^3 sediment. Within a site, however, the range in microplastic density was also large, with an average range across nine samples of 0.27 plastics per cm^3 of sediment, and a maximum range across nine samples of 1.66 plastics per cm^3 of sediment at Saba Bay (coefficient of variation = 2.53). The size range of plastics was 0.5 mm (lower limit) to objects larger than 10 cm.

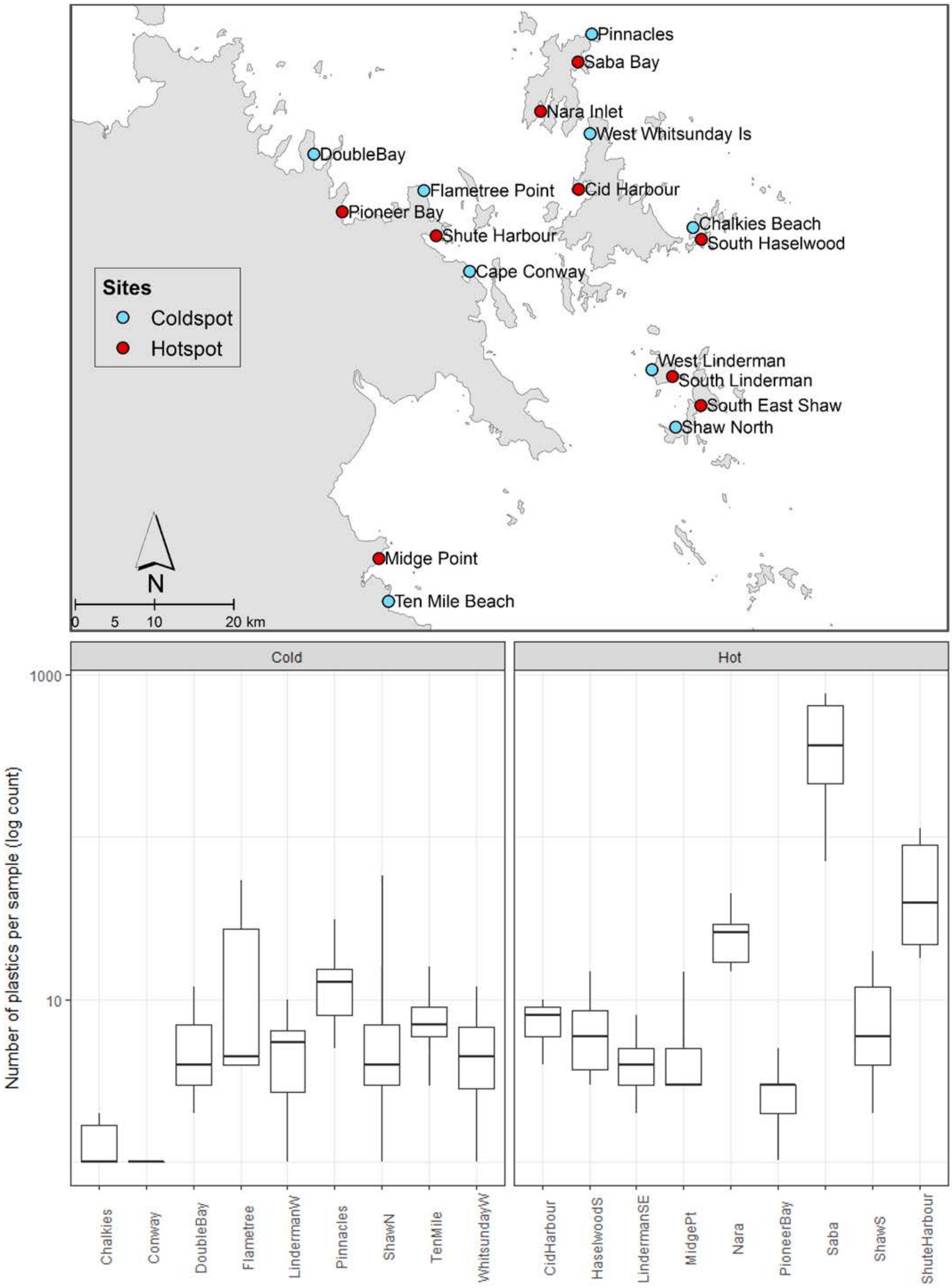


Figure 4.3: *Number of plastics found at each of the sampling sites (top panel). Data presented in the bottom panel are the number of plastic particles per sample - boxplots present the median mid-line of the boxes, the 25th and 75th percentile (box limits) and the vertical lines represent the range.*

4.3.3 Macroplastics field data overview

Macroplastic debris loads, and accumulation rates, were highly variable among sites. The maximum number of total items removed from a site in one visit was 17,000 items (Saba Bay) compared with the minimum number of 56 items removed from Hamilton Island (South End of Runway; Figure 4.4). Accumulation rates were also highly variable among sites, ranging from 3 items per week at Pine Bay to 725 items per week at Turtle Bay. There was no strong spatial pattern to macroplastic loads with high values observed on both land-facing and ocean-facing beaches (e.g., Whitsunday Is.; Saba Bay; Figure 4.4).

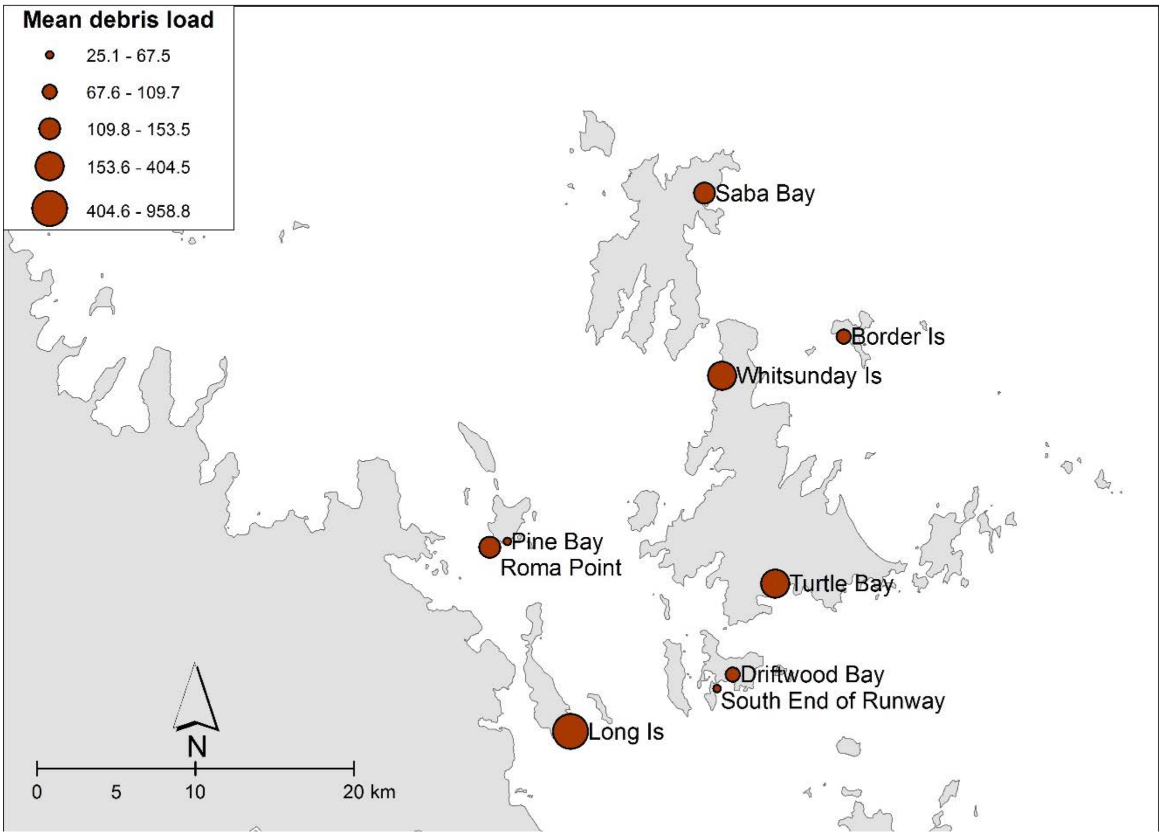


Figure 4.4: *Mean debris loads from all collection visits to macroplastic debris removal sites. The mean debris load is depicted by the size of the red circle.*

4.3.4 Source of the plastic to the region

Microplastics

It would be expected that the scenarios using wind data closest to the field sampling (June or August scenarios) would most closely match the field data, however, simulations implemented using different seasonal winds suggest that there are different sources of plastics arriving in different seasons (Figure 4.5). Using February winds, sites were most accurately predicted as hotspots and coldspots in simulations that included only marine sources (67% of sites accurately categorised, Figure 4.5). Plus, when examining the dispersal plumes from the marine and land-based source locations after the 45 day simulation (Figure 4.6), it can be seen that very few particles from the land-based sources reach the northern and eastern islands during the period of simulation. Conversely, very few of the particles from marine sources reach the mainland during this same simulation. In the February set of simulations, model predictions based on particles from the combined sources fit poorly with the observed data (Pearsons R^2 = 0.24, p-value = 0.038, 56% prediction accuracy Figure 4.5). However, when the scenario was split into marine and land-based sources, the fit between the observed and predicted accumulation was improved for the marine source component of the model (Pearsons R^2 = 0.56, p-value = 0.00037, Figure 4.6). It is therefore unlikely that land-based sources supply microplastics to the field collection sites under the conditions experienced in February.

In contrast, simulations using August and March winds better predicted hotspots and coldspots using only plastics derived from land-based sources (56 – 67% of sites accurately categorised, Figure 4.5). Moreover, using winds from June, combining both land-based and marine sources categorised the most sites correctly (56% of sites correct, Figure 4.5). Although prediction success was generally low (maximum of 67% sites correct), the categorisation of hotspots and coldspots was more accurate than the prediction of the actual values of plastics present at each site (Figure 4.6). The accumulation correlations for only the February scenario are shown in Figure 4.6, the accumulation correlation graphics and statistics for the other scenarios can be found in Appendix 2: these have lower correlation values for all scenarios.

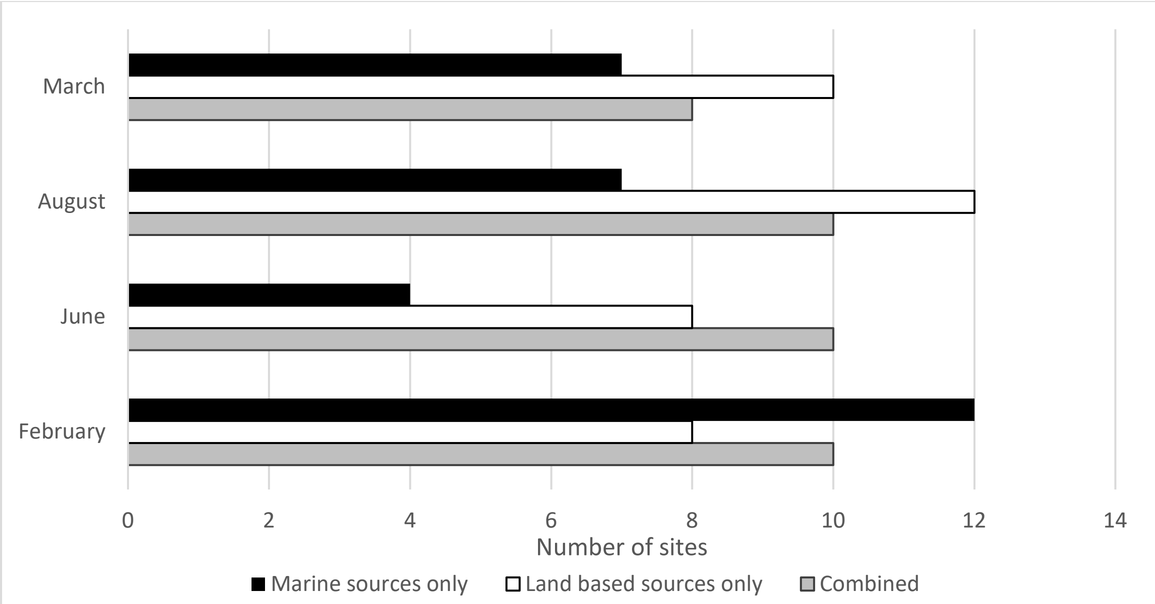


Figure 4.5: *The number of sites correctly categorised by each scenario. For each of the scenarios, the sites were re-categorised based on the median predicted accumulation value for that scenario. Higher than the median values were designated "predicted hotspots" and lower than the median values were classified to be "predicted coldspots".*

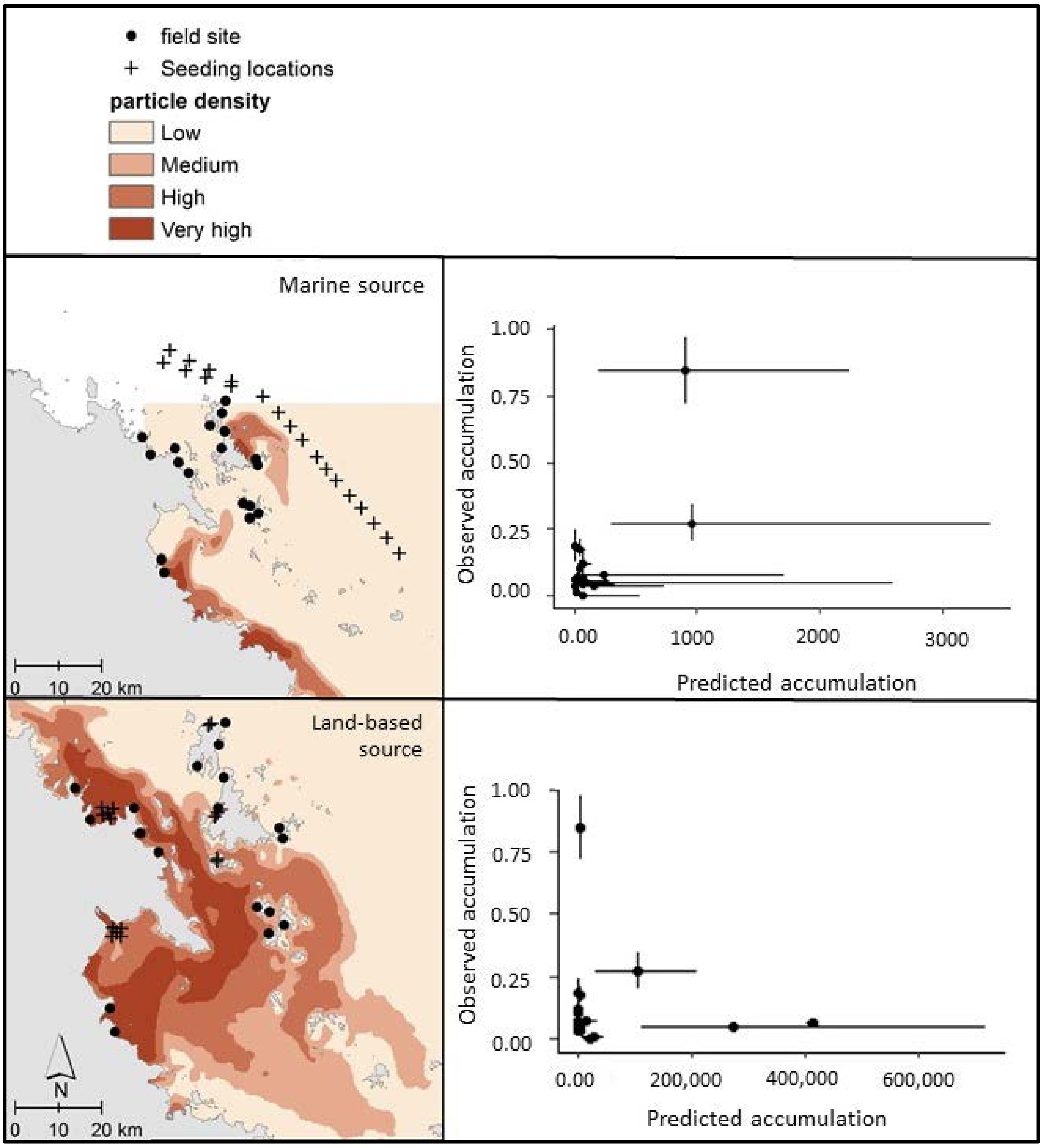


Figure 4.6: *Particle distributions for microplastics from land-based and marine sources during the February scenario. The left panel shows the plumes of the two broad source locations at the end of the February 45 day scenario, for microplastics. The right panel shows the fit of the observed and predicted accumulation values at the field sites, the y axis error bars represent the standard error and the x axis error bars show the 25th and 75th quartiles of the daily predicted accumulation values through the simulation.*

Macroplastics

Among all the scenarios (Table 4.1), the original scenario categorised the most sites correctly with all nine sites correctly categorised as macroplastic hotspots. Unlike the microplastics model, in all macroplastic scenarios the combined source model (pooling both land-based and marine sources) performed at least as well as models separating land and marine sources (Figure 4.7). For the macroplastic data, there was no scenario in which predictions based solely on marine-based sources had the highest hotspot/coldspot prediction success.

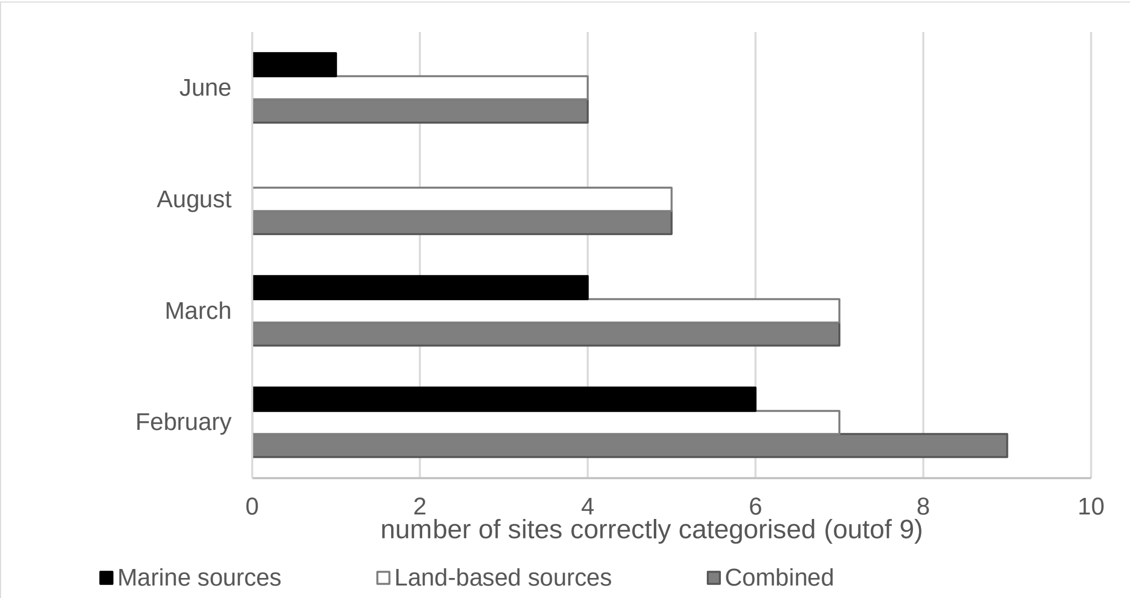


Figure 4.7: *The number of sites correctly predicted in each scenario.*

The predicted accumulation values for individual sources did not correlate with the observed accumulation (Figure 4.8), and did not perform better than the combined-source model (Appendix 2). In comparison to the microplastics model for February (Figure 4.6), the dispersal plume of marine sources for macroplastics is not so effectively blocked by the islands, as the dominant movement process is driven by the wind. Therefore, the coastal sites receive simulated macroplastics from the marine sources (Figure 4.8), and the macroplastics from land-based sources are more likely to beach on the coastal locations (Figure 4.8) than the microplastics (Figure 4.6). Indeed, in this scenario, macroplastics from land-based sources were unable to disperse to sites >20 km away from the mainland (Figure 4.8).

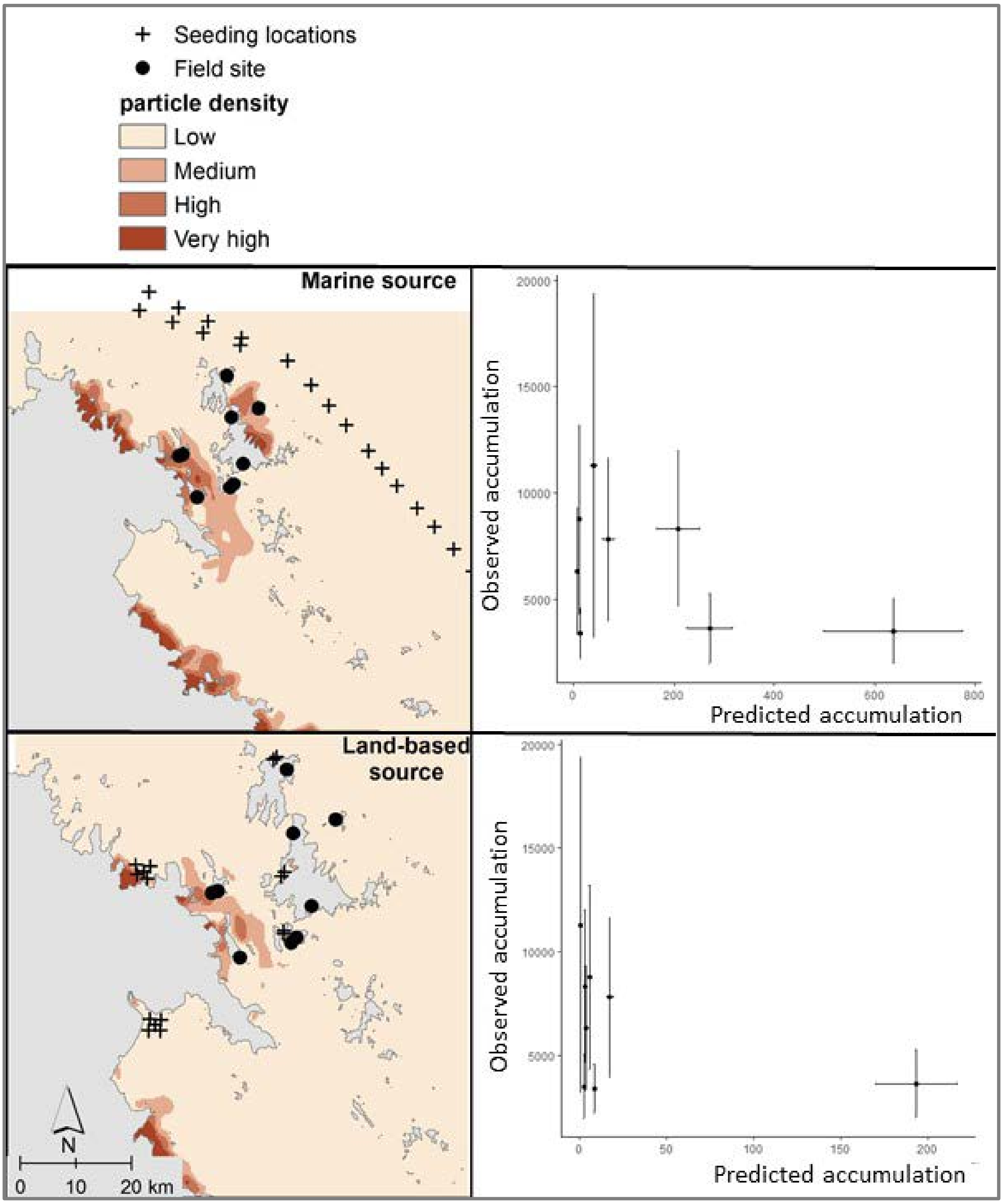


Figure 4.8: *Particle distributions for macroplastics from land-based and marine sources for the February scenario. The left panel shows the plumes of the two general source locations at the end of the February scenario, for macroplastics. The right panel shows fit of the observed and predicted accumulation values at the field sites.*

4.3.5 Non-source parameters

To explore how processes other than source location influence plastic accumulation, I changed the source locations from point source type seeding of the land-based and marine sources to a grid of uniform seeding locations representing a ubiquitous load of plastic occurring across the whole study area (Table 4.1; Figure 4.9A). The microplastic accumulation areas predicted by the grid seeding scenario correctly categorised 12 of the 18 field data sites as either hotspots or coldspots (66.7% prediction success), and improved the fit between the observed and predicted accumulation values, but still did not result in a statistically significant correlation between predicted and observed accumulation (Pearsons $R^2 = 0.03$, p-value = 0.49, Figure 4.9B).

Using the same process as for the microplastics, I used the grid seeding scenario to explore the dominant process for macroplastics. The grid seeding for macroplastics correctly categorised six of the nine sites (66.7% prediction success). However, the relationship was not as strong as for microplastics. The predicted accumulation values for both macro- and microplastics from the grid scenario do not correlate well with the field data and many sites have predicted accumulation values of zero (Figure 4.9C).

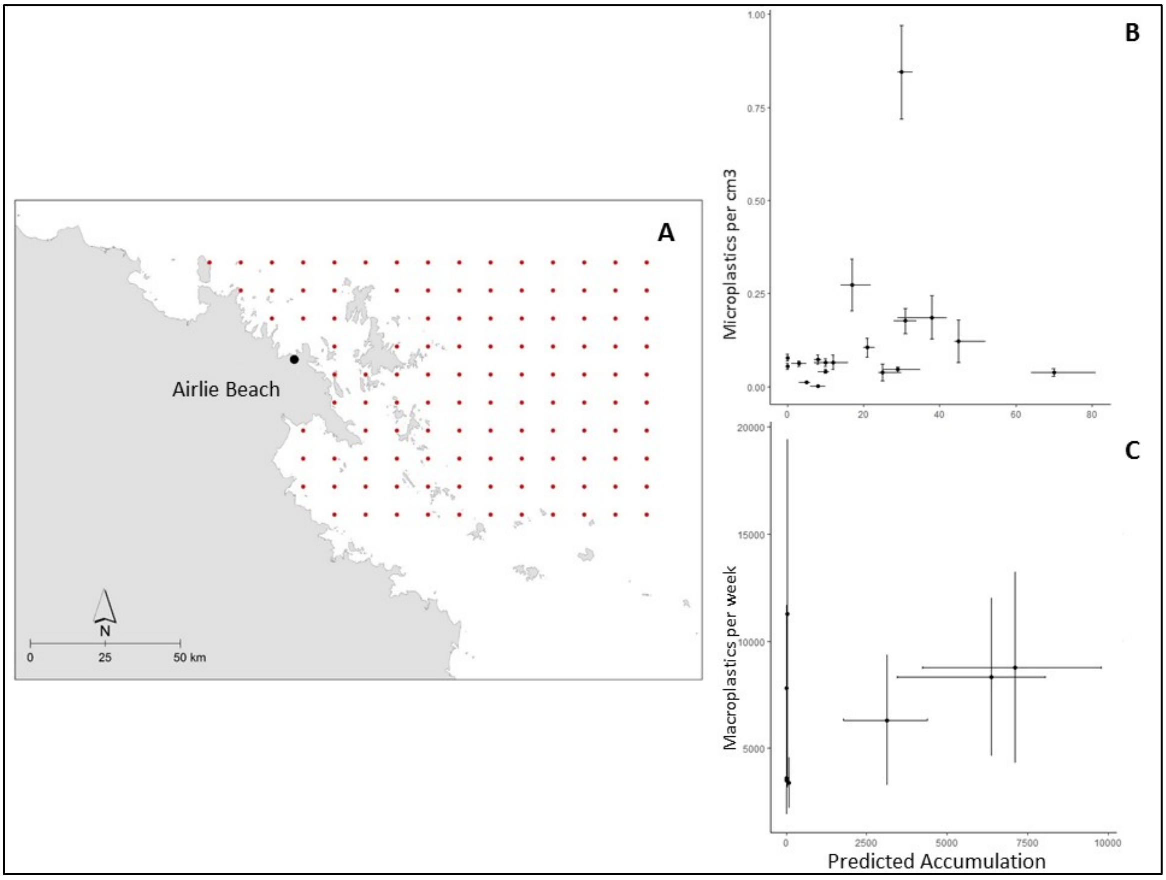


Figure 4.9: *Details of the grid seeding scenario: A) seeding locations used in this scenario, B) scatter plot showing the observed and predicted accumulation values of the microplastic scenario, and C) scatter plot of the observed and predicted accumulation per week of macroplastic scenario. In both scatter plots the y axis error bars represent the standard error and the x axis error bars show the 25th and 75th quartiles of the daily predicted accumulation values through the simulation.*

4.3.6 Characteristics of sites that influence prediction success for plastic accumulation

The sites that the model most often predicted correctly to be microplastic hotspots or coldspots were; Pioneer Bay (coldspot), Shute Harbour (hotspot), and Saba Bay (hotspot), which were all correctly categorised in 8 of the 12 scenarios (66.7% success rate). Conversely, Ten Mile Beach was only predicted correctly by one scenario (Figure 4.10). The sites with a high rate of prediction success don't share any obvious physical characteristics, for example, bay shape and coastline orientation are different for each site and there was no indication that proximity to the mainland influenced prediction success (Figure 4.10). The mean prediction success was below 50% for both open and closed bays (42.7% and 47.5%, respectively) as well as for east- (towards open ocean) and west- (towards the coast) facing sites (44.2% and 46.9%, respectively).

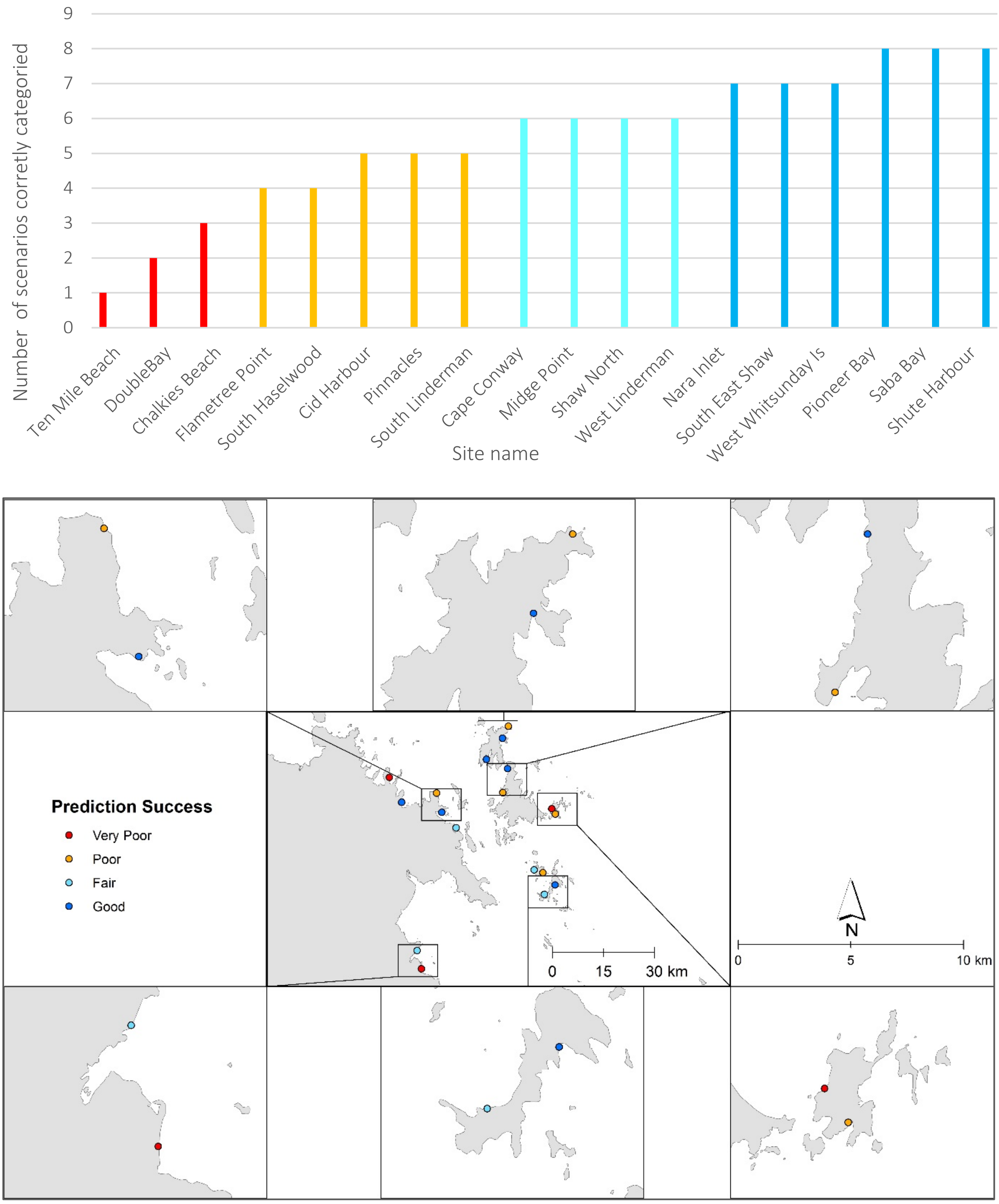


Figure 4.10: *The predictability of each microplastic field site (i.e. the ability to categorise the site as hot or coldspot correctly). The top panel shows the number of times each site is correctly categorised in any scenario and bottom panel shows the geographic spread of predictive success. Insets show case studies of sites with various success rates. Success categories are coloured Blue ("Good") to Red ("Very Poor") scale shown in the legend and in the graph.*

Prediction success was not correlated with REI (Pearsons $R^2 = 0.004$, p-value = 0.791, Figure 4.11A). Neither was REI correlated with the number of microplastic particles found at the sites (Pearsons $R^2 = 0.024$, p-value = 0.539, Figure 4.11B).

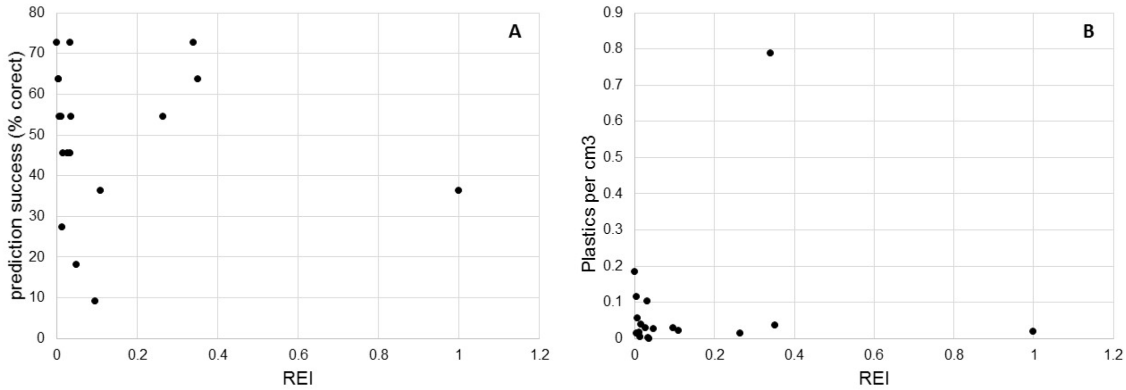


Figure 4.11: *The correlation between A) relative exposure index and the prediction success of each microplastic site, and B) the relative exposure index and the observed accumulation at each microplastic site.*

4.3.7 The site-specific processes that result in accumulation of macroplastics

As with the microplastics, I assessed the number of times any scenario correctly predicted the observed accumulation category (hotspots only). The sites most often predicted correctly were; Saba Bay (91.7% prediction success), Turtle Bay and Runway (both 66.7% prediction success). Conversely, Border Island was only correctly categorised in two of the 12 scenarios (Figure 4.12). As for microplastics, the sites with high success do not seem to share physical characteristics, such as shape and orientation.

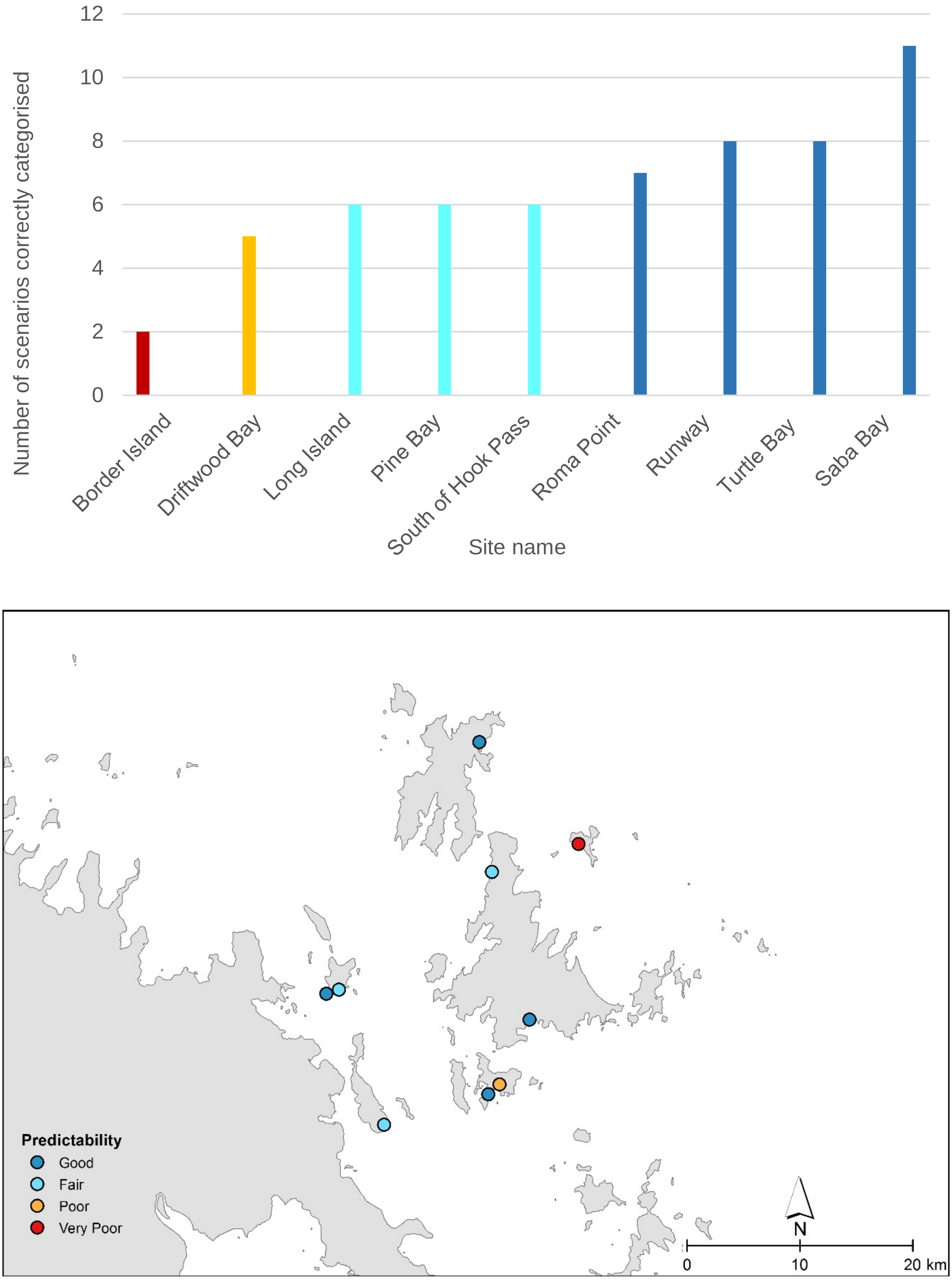


Figure 4.12: *Prediction success of macroplastic sites, bar graph shows the number of times each site is correctly categorised in any scenario. The map shows the geographic spread of predictive success. Success categories are coloured Blue ("Good") to Red ("Very Poor") scale shown in the legend and in the graph.*

To explore the role of exposure in the accumulation at each site, I compared the relative exposure index (REI) with the prediction success of each site and the observed accumulation per week at each site (Figure 4.13). There was no significant correlation with the prediction success (Pearsons $R^2 = 0.0388$, p-value = 0.611). While sites that had higher REI values generally have higher observed accumulation values, this relationship was not statistically significant, most likely due to small sample size in the field data (Pearsons $R^2 = 0.32$, p-value = 0.112). This warrants further study.

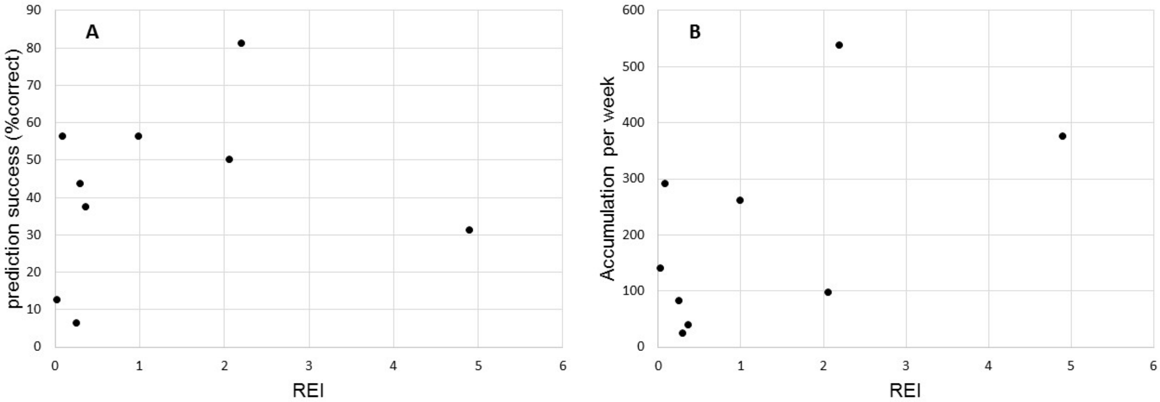


Figure 4.13: *The correlation between A) relative exposure index and the prediction success of each macroplastic site, and B) the relative exposure index and the observed accumulation at each macroplastic site.*

4.4 Discussion

In this chapter I used field data to interrogate the SLIM plastics hydrodynamic model to better understand the processes controlling the dispersal and accumulation of plastic pollution. I found that the most likely sources of the plastic accumulating on beaches around the region are different for macro- and microplastics, and that the dominant sources contributing plastic to local beaches differs among seasons. Depending on differences in seasonal winds, macro-plastics from land-based sources can be prevented from dispersing away from the mainland, and differences in seasonal currents prevented microplastics from land-based sources from reaching certain sites on offshore islands. In contrast, the presence of offshore islands blocked the dispersal of both micro- and macroplastics from marine sources, with plastics from these sources rarely reaching the leeward (mainland-facing) side of these islands. Nevertheless, observed spatial patterns of macroplastics accumulation rates, and microplastic abundances, diverged from model predictions. I found no strong evidence that differences between model predictions and field observations were driven by factors like distance from the mainland, beach orientation or bay shape but, for macroplastics, correct prediction of debris hotspots was positively (although not significantly) associated with the exposure of beaches. These results suggest that coupling a model for dynamics of wind-driven waves with the SLIM system could be useful. However, hotspot/coldspot prediction success also depended on seasonal winds which indicates that knowing the timing of arrival of particles on beaches (and/or timing of release of particle from sources) is important for accurately predicting plastic dispersal pathways. Thus, obtaining data on the arrival and persistence of micro- and macroplastics at beaches, while challenging to collect, would almost certainly improve the statistical power of future modelling exercises.

Source is one of the most important parameters to understand when attempting to model plastic pollution at a small scale in coastal environments (see Chapter 2). Macroplastics degrade slowly in seawater (Andrady, 2011), and are released into the environment in a similar state to the state that they are found on the beach. Indeed, Carson et al., (2013) used drifters to demonstrate that 23% percent of plastics released from local rivers in the Hawaiian island of Hilo were found on local beaches. Interestingly, the authors also found low agreement between the modelling outputs and the field data, and stressed the importance of nearshore tidal dynamics in modelling debris in the coastal zone. In Australia, Reisser et al., (2013) conducted a study using field data and known sources and concluded that microplastics on the east coast of Australia are derived from local sources. However, the samples were collected off the coast adjacent to major cities rather than in more remote areas with a relatively low population density such as occurs in my study area. Quantifying specific sources for the micro- and macroplastic beaching in the

Whitsundays region is a new challenge. My data indicate there is a correlation between accumulation and exposure for macroplastics, inferring the dispersal of macroplastics is influenced by wind speed and direction, and suggests a need to couple wind-driven waves into hydrodynamic models of plastic dispersal. In the Whitsundays region, there are two distinct wind seasons, the strong south-easterly trade wind season (corresponding to the seasonal winds used in my model predictions) and the monsoon season which is characterised by weaker more variable northerly winds (Wolanski et al., 1981, also see Figure 3.2). Given that the macroplastics data analysed in my study were collected in both of these wind seasons, and that my results show that prediction success for macroplastic hotspots depends on winds, future debris dispersal simulation should be conducted under a broader range of wind conditions. Moreover, the hotspots for macroplastic accumulation are likely to differ across seasons which could have implications for the timing and locations of beach clean-up or other mitigation efforts.

Overall my data suggest that measuring the movement and accumulation of macroplastics and microplastics in the coastal zone are more complex that they first appear. Microplastics have been shown to be patchy in time and space (Reisser et al., 2013), and the physical location of accumulation varies at the timescale of season and day (Ryan et al., 2014; Hu et al., 2016). For example, buoyant macro- and microplastics have been shown to accumulate in convergence zones and fronts (Acha et al., 2003; Lima et al., 2015) and sampling efforts not considering, or measuring the presence of the fronts could under- or overestimate environmental plastic loads. At small spatial and temporal scales, the tides and wind both move convergence zones at a scale of hundreds of meters (or less) (Hu et al., 2016). Plastic polymers also have varying densities and this will also influence how the plastic object is transported (Chubarenko et al., 2018), however, the shape of the object can overcome this, for example, if an object is hollow or with air pockets the buoyancy of the object as a whole will be different again. All these sources of variability would reduce the capacity to accurately predict the accumulation of plastics in the ocean and on beaches.

Although wind is clearly important, the lack of a strong relationship between accumulation areas with exposure to wind indicates there are other attributes that were not included in the exposure model framework. The field data for the microplastics represent the net accumulation of plastic waste over an unknown period of time and, for the macroplastics, the field data at each site represents the net accumulation over different time periods. Therefore, the observed plastics accumulated during unknown and variable winds and currents that are likely a result of a mix of environmental conditions potentially not displayed in the model. A more specific test of the model predictions would require forward modelling of dispersal from specific source locations (as conducted in this study) combined with satellite tracked drifters released from those same locations and I suggest that such an approach could enable more specific refinements of hydrodynamic models that aim to predict plastic dispersal. It is also possible that there are missing source locations in this study, such as unknown offshore sources, reducing the ability of the model to accurately reproduce observed accumulation. Parameter values calculated from field data or experiments alongside more field data would be necessary to understand these additional factors. Given the unknown timeframes the plastics accumulated on the beaches, I also suggest collecting specific macroplastic data at the same time and location as the microplastic samples, or to design an experimental setup to trap microplastics as they wash ashore, or are moved along a beach to give more accurate data on their arrival and departure from the beaches. It is also likely that collecting a time series of field data across many wind and wave scenarios would lead to a more robust model analysis. Future studies should consider assessing all plastic types at each site that they visit, and through time. The timing of sampling has also been shown to be important (Smith and Markic, 2013), which should been taken into account during studies of this kind.

A common theme in the results of this chapter is high variability, both in the plastic accumulation data and in the modelling results. In addition, it is likely that the plastics observed on the beach (both microplastics and macroplastics) represent accumulation since the last major weather event that could resuspend and export the plastics from the beach, back into the water column. The type of weather event that could export the plastics is different for macroplastics and microplastics. For example, buoyant macroplastics on the coastline are easily moved by waves and tides along shore and it is the strength of the offshore processes acting at the convergence point that dictate the export of buoyant macroplastics from the beach to the sea (Kataoka et al., 2015; Hinata et al., 2017). These buoyant objects have high upward velocity, however, microplastics have very low upward velocity due to their smaller size. Therefore, unless microplastics are buried in the sediment at a depth below the level waves have influence on the sediment mixing, the waves will move microplastics easily offshore and they are then generally exported from the near-shore area. This process was described by Jackson et al., (2014) for the eggs of horseshoe crabs in beach sediment, which would likely behave in a similar way to microplastics because they are less dense then the sediment they are trapped in. There have been other analysis of residence time on beaches, among them, the study by Hinata et al., (2017) regarding the residence time of microplastics on beaches derived from the residence time of macroplastics. The conclusion of the study is likely inaccurate, because the authors consider the upward velocity of microplastics to be equivalent to that of macroplastics. This is not the case, Stokes Law shows that as the diameter of a particle decreases the upward velocity decreases (shown in Reisser et al., 2015). For microplastics the upward velocity would be negligible at the time-scale of a wave and would therefore be moved in a very passive way.

Another aspect to consider is the physical attributes of the site that is accumulating plastics. There was no consistency in physical features of the sites found to have highest accumulation in the field data, for macroplastics or microplastics. Orientation and bay shape were expected to be important in predicting accumulation, however, these factors did not consistently influence macroplastic accumulation rates or microplastics abundances, nor did they affect the prediction success of plastic hotspots and coldspots. This may mean that many bay shape/orientation combinations will increase the likelihood of a site being an accumulation site but there were not enough examples of each in the data set available here to understand the patterns or draw significant statistical power from my results. Small topographic features may also influence the accumulation as they create localised sub-beach level convergence that would create patchy accumulation easily missed by the field sampling (Wolanski and Hamner, 1988; Kataoka et al., 2015). Similarly, on beaches the rates of sand accretion, deposition and sand grain size are not uniform along a single beach, and it is possible that small pockets of sand/plastic accretion occur in relation to beach specific factors such as beach shape (for example, the slope or presence of ridges), vegetation, tidal reach (Dawson and Smithers, 2010). More field sites and more samples per beach are necessary to understand these processes in more detail, or a comprehensive study of a single beach at a very fine spatial resolution. I also suggest separate models be developed for each plastic type, using parameter values specific to their type or object, such as the effect of wind-drift (e.g. Kako et al., 2010), as different objects and polymer densities can display very different transport behaviours. These plastic-type specific models could then be combined to understand the variability of specific types of plastic pollution as a whole in a particular area.

There are some limitations to the study presented in this chapter. Firstly, the advection time (or integration time) has been found to be a very important factor in the modelling of dispersive objects (Mansui et al., 2015). All my simulation lengths were chosen to be comparable to one another, but if I had conducted simulations of different durations the results would have differed. The results of Mansui et al., (2015) show a marked difference in the model output after three months compared to one year. However, the results are not directly applicable to my study area, because the study of Mansui et al. was conducted at the scale of the whole Mediterranean Sea which is considered to be a semi-closed to closed system. The Whitsundays region of the Great Barrier Reef is much smaller and could be considered a semi-open system because water can flush though the study area in either direction. In future projects, multiple time scales should be considered, by incorporating multiple release times throughout the simulation, and running the scenario for much longer periods to see if there is indeed convergence to a stable spatial distribution of macro- and microplastic particles. Another limitation of this study was in using the median value as the threshold when reclassifying the sites into hotspot and coldspots for both the observed and predicted accumulation. Using the median value may have resulted in misclassification of some sites. This is especially important in comparison to the method used to select hotspot and coldspot locations for field sampling. The original classification for coldspots was that the predicted accumulation had to be nil to very low; by using the median value to reclassify there may have been sites that were reclassified as coldspots that should have been hotspots as they had some accumulation, i.e. if I had used the nil to very low threshold, sites with some accumulation may have been classed as hotspots. A consensus in the literature on what constitutes a hotspot would be useful for future studies of this kind. The use of the median value for sites and datasets that have such high variability may also misrepresent the correlation of observed and predicted accumulation values.

Overall, this chapter shows that microplastics and macroplastics should not be treated as equivalent in modelling or movement studies. The outlook for modelling microplastic dispersal and accumulation is promising and the SLIM performed well to consider the physical properties of dispersal. However, it is clear from my analysis that more field-based data on beached plastic is necessary to improve the correlations between modelled and field data and to train the model. That said, my results provide one of the first empirical studies to compare field and modelled data of both macro- and microplastics in a coastal environment. Collectively, my results improve our understanding of the sources and physical processes that dictate plastic accumulation in the coastal zone, and will ultimately improve our ability to predict accumulation hotspots through modelling and allow them to be used as an empirical base for municipal-scale management action. Understanding sources is important because management strategies will differ based on the source. If the source is local, the council can enact local interventions that have noticeable results on local habitats. However, if the source is external, the local council must coordinate efforts with other councils or the state to reduce inputs (Smith, et al., 2014). Finally, obtaining a strong correlation between observed and predicted data would be a strong benefit to management action and this chapter provides a framework for achieving that in the future.

Chapter 5
Effects of microplastic exposure on the body condition and behaviour of planktivorous reef fish (*Acanthochromis polyacanthus*)

The effect of a pollutant on the base of the food web can have knock-on effects for trophic structure and ecosystem functioning. In this study I assess the effect of microplastic exposure on juveniles of a planktivorous fish (*Acanthochromis polyacanthus*), a species that is widespread and abundant on Indo-Pacific coral reefs. Under five different plastic concentrations, with plastics the same size as the natural food particles (mean 2 mm diameter), there was no significant effect of plastic exposure on fish growth, body condition or behaviour. Consumption rates were low, with a range of one to eight particles remaining in the gut of individual fish at the end of a 6-week plastic-exposure period, suggesting that these fish are able to detect and avoid ingesting microplastics in this size range. However, the number of plastics found vastly increased when plastic particle size was reduced to approximately one quarter the size of the food particles, with a maximum of 2102 small (< 300 µm diameter) particles present in the gut of individual fish after a 1-week plastic exposure period. Under conditions where food was replaced by plastic, there was a negative effect on the growth and body condition of the fish. These results suggest plastics could become more of a problem as they breakup into smaller size classes, and that environmental changes that lead to a decrease in plankton concentrations would likely have a greater influence on fish populations than microplastic presence alone.

Citation: Critchell, K., and Hoogenboom, M.O., in review *"Effects of microplastic exposure on the body condition and behaviour of planktivorous reef fish (Acanthochromis polyacanthus)" PlosONE*

5.1 Introduction

Plastic pollution has been reported in every ocean and sea on Earth (Eriksen et al., 2014), and is widely recognised as a global threat to marine life, and to the economies of coastal nations (Derraik, 2002; Thompson et al., 2009). Plastic pollution can degrade coastal benthic habitats through smothering, when plastic sheets form a layer over the benthos and, also, through changes in sediment permeability due to buried plastics (Carson et al., 2011). Moreover, plastic pollution causes harm to wildlife through entanglement and ingestion (Derraik, 2002). Although plastics do not readily biodegrade, they do break up into smaller pieces when exposed to ultraviolet light and physical abrasion (Andrady, 2011). As plastic particles become smaller they become available to be inadvertently consumed by a wide range of marine organisms. Ingestion of plastics has been reported in many species of marine fauna, most notably seabirds (Kinan and Cousins, 2000; Verlis et al., 2013) and sea turtles (González Carman et al., 2014), but also fish (Hoss and Settle, 1989; Possatto et al., 2011; Neves et al., 2015), corals (Hall et al., 2015), and other invertebrates (Wright et al., 2013a; Setälä et al., 2014; Van Cauwenberghe et al., 2015). Consumption of microplastic by organisms at the base of food webs, such as mussels (Farrell and Nelson, 2013) and plankton (Cole et al., 2013), has raised concerns about the potential for transfer of plastic-associated toxins throughout marine food webs (Thompson et al., 2009).

Ingested plastics can cause harm though physical damage to the gastrointestinal (GI) tract. For instance, plastic ingestion can cause abrasions and lesions, or physical disruption of the GI tract, as plastics compact in the gut (e.g. Di Bello et al., 2013). Plastic ingestion can also be detrimental to the health of various organisms because indigestible particles fill the stomach and reduce the feeling of hunger which leads to starvation (e.g. Ryan, 1988; Spear et al., 1995). Conversely, microplastics (defined as plastic fragments < 5 mm) have been found in the GI tract of marine animals without causing obvious harm. For example, 12.2% of harbour seals assessed in the Netherlands contained microplastics and, there was no clear effect of the plastic consumption on the animals (Bravo Rebolledo et al., 2013). However, many studies to date have simply reported the presence of plastics in the GI tract without assessing the effects on the fitness of the organism (Cole et al., 2013; Watts et al., 2014). Overall, while there is growing evidence that many different taxa consume microplastic particles, the potential health effects of such ingestion are not well known.

In addition to the physical effects of plastics on animal digestion, many plastics contain chemicals, such as flame retardants and plasticisers, which are added during the production process to give the plastics certain properties (Derraik, 2002; Rochman et al., 2013a; Eerkes-

Medrano et al., 2015). Plastic additives can be transferred into the tissues of animals that have consumed plastics (Rochman et al., 2013b; Chua et al., 2014), with potential effects on the physiology and health of the animal. Laboratory experiments have been used to assess the effect of plastic consumption on the development of oganisms. For example, lugworms in an experimental trial were found to lose weight over a 28 day exposure to microplastics with polychlorinated biphenyls (PCBs - Besseling et al., 2013), but it remains unclear whether it is the physical presence of microplastics, or the toxic effects of PCBs, that cause this effect. PCBs can alter the regulation of key hormones including oestrogen, testosterone and thyroxine (Colborn et al., 1993; Goncharov A et al., 2009). Changes in hormone concentrations can have complex effects on animal behaviour. For example, increased testosterone levels are associated with behavioural dominance in fish and mammals in general (Hirschenhauser and Oliveira, 2006). Independent of toxicological effects, changes in behaviour can also be expected in response to starvation. For instance, a hungry individual may become more aggressive or listless; it may also increase territoriality (e.g. Adams et al., 1995; Dunbrack et al., 1996; Adams et al., 1998). Despite the important role of animal behaviour in determining performance in the natural environment, the effects of microplastic exposure and/or ingestion on animal behaviour remain largely unknown.

Although plastic consumption by marine vertebrates is arguably best documented in seabirds (e.g. Ryan, 1987), many studies have also demonstrated that teleost fish consume microplastics in the natural environment (Vendel et al., 2017; Hoss and Settle, 1989; Possatto et al., 2011; Foekema et al., 2013; Rummel et al., 2016). For seabirds, up to 80% of individuals of some species are reported to contain plastics (Acampora et al., 2014), and the mortality from starvation is obvious when seabird carcasses are observed on beaches (e.g. Pierce et al., 2004). For fish, field studies have revealed that up to 30% of individuals have plastics in their GI tracts (Romeo et al., 2015; Rummel et al., 2016). However, some species of fish generally retain lower numbers of plastic particles per fish (two to four) (Romeo et al., 2015; Rummel et al., 2016). Moreover, plastics consumed by fish tend to be small in size, with 22 of 121 fish gut-content samples from the Mediterranean containing plastics of which 70% were < 5 mm (Romeo et al., 2015). Nevertheless, it is currently unknown whether ingestion of small quantities of microplastics is detrimental to the health of these fish. To date, field studies of fish plastic ingestion have primarily focused on pelagic and commercially important species, including mackerel and cod (Foekema et al., 2013). These fish prey on smaller species of fish and it is unclear whether these piscivorous fishes consume microplastics directly from the water column or whether they incidentally ingest plastics by consuming prey that had eaten plastics themselves. Understanding the level of fish stock contamination by plastics requires an understanding of these trophic links, and knowledge of whether the small plantivorous fish, that are the prey of fisheries species, consume and retain microplastics.

There is marked variation in fish diets, both within and among species. This difference stems from feeding strategy or natural prey size differences (O'Brien et al., 1976; Ma et al., 2015). Some species have a highly selective diet (Hobson and Chess, 1977; Ma et al., 2015) suggesting that such species might only rarely eat plastics in their natural environments. Carpenter et al., (1972) showed that white spheres were the only microplastics found in the gut of eight different fish species, indicating selective feeding of white spheres over other types of plastics. Fish also display ontogenetic changes in diet as they grow larger, with smaller fish generally eating smaller prey (e.g. García-Berthou, 1999). Therefore, fish of different sizes might be more likely to eat a particular size range of plastics. For example, some damselfish species increase their reliance on consumption of benthic algae as they mature (Emery, 1973), meaning that juvenile fish that consume plankton may be more at risk of harm from microplastic consumption than adults. In addition to ontogenetic shifts in diet, fish show individual variation in their feeding behaviour (e.g. Hoogenboom et al., 2013) and might differ in their propensity to consume plastics. Moreover, fish show high variation in their responses to toxins (e.g. Schwaiger et al., 1997; Jaffal et al., 2015). Understanding effects of plastic ingestion on fish populations therefore requires quantification of among-individual variation in the propensity to ingest plastic, and among-individual variation in the effects of plastic ingestion.

The effects of plastic exposure on fish growth and behaviour are likely to be concentration-dependent. If plastic consumption depends on plastic availability in seawater, greater plastic ingestion, and greater potential impacts of plastic feeding, should be observed at higher plastic concentrations. In an extreme case, if plastic concentrations become so high that they replace plankton in seawater, then this is likely to alter fish growth and behaviour through starvation effects. In this study, I aimed to quantify whether and how plastic ingestion by fish depends upon plastic concentration, and the effects of plastic consumption on the growth and body condition of juvenile planktivorous reef fish. I also aimed to determine whether plastic consumption and/or plastic presence in seawater affected fish behaviour. This is to test the hypothesis that an increase in the concentration of a poor food resource (plastics) may lead to an increase in aggressive behaviour due to more frequent failed foraging efforts. My final aim was to assess the likelihood of plastic consumption with different size classes of plastics for different size classes of fishes. It was my hypothesis that reduced plastics size would increase the rate of consumption as the ability to distinguish between food and non-food particles was reduced.

5.2 Methods

5.2.1 Ethics Statement

This study was carried out in strict accordance with ethics protocol laid out by the *Animal Ethics Committee of James Cook University,* who approved the protocol of these experiments (Permit number: A2112), and priority was given to animal care at all stages of this study.

5.2.2 Overview of approach

An experimental approach was used to isolate the effects of plastic consumption for a common planktivorous reef fish under otherwise controlled conditions. I chose juvenile *Acanthochromis polyacanthus* as a representative species. *A. polyacanthus* is a geographically widespread planktivorous reef fish, common throughout the Great Barrier Reef (GBR), and like most planktivores, has relatively low feeding selectivity (Emery, 1973), making them a good candidate for feeding trials. Also, *A. polyacanthus* is a commonly used experimental fish species as they are easy to rear and care for in a laboratory environment.

I conducted an aquarium-based experiment to determine whether plastic ingestion affected the growth and body condition of juvenile *A. polyacanthus*, and whether any effects were concentration dependent. This experiment consisted of two phases. First, I assessed whether replacement of food by plastic was detrimental to fish growth (referred to hereafter as 'acute exposure'). Second, I assessed the influence of exposure to plastics in addition to the normal level of food (referred to hereafter as 'chronic exposure'). To minimise the number of animals exposed to plastics (consistent with animal ethics guidelines), these phases were implemented sequentially, with one week of acute exposure followed by six weeks of chronic exposure. Growth rates (length and mass) were measured weekly. A separate aquarium experiment was conducted to assess whether ingestion rates of plastic by fish depend on plastic particle size. In that experiment I exposed fish of two size classes to three size classes of plastic particles for one week, after which their gut content was analysed to assess the amount of plastic retained.

Experiments were conducted in a large recirculated marine aquarium system at the Marine and Aquaculture Research Facility (MARF) at James Cook University. Water in aquaria was maintained at a temperature of 27.5 °C (±1 °C), natural pH, salinity of 36 ppt, and with nitrates (NO_3) within the range 28.5 and 32 mg l^{-1}. Tank internal dimensions were 470x325x280 mm (Figure 5.1). Juvenile *A. polyacanthus* (N = 112) from three sets of parent stock (hereafter, 'clutches', labelled A-C) that had been raised in captivity at MARF, were reared between February and April 2015. Juvenile fish were raised at low stock densities and were fed a commercial, high nutrition food twice per day *ad libitum* until the cohort average length (fork length) was approximately 3.5 cm. The fish from each clutch were randomly split into one of five treatments with different microplastic concentrations: control (0 mg l^{-1} plastic), low (average 0.025 mg l^{-1} plastic), medium (average 0.055 mg l^{-1} plastic), high (average 0.083 mg l^{-1} plastic) and very high (average 0.1 mg l^{-1} plastic). There were two replicate aquaria per clutch, per treatment for a total of six replicate aquaria per treatment. During the experiment, the fish were fed the same type of commercial food pellets as the growth phase and food amount was adjusted according to fish biomass in the aquaria. The amount of plastic added to tanks was also adjusted according to fish biomass in the aquaria, so that there was a constant provision of plastic per unit fish biomass in each treatment. Differences in fish biomass among clutches, and over time, meant that the amount of plastics provided to tanks overlapped slightly between treatments (see Appendix 3, Table 1). Fish were fed twice a day for the duration of the experiments and were left to feed for at least four hours after exposure. After four hours, the aquaria were cleaned to remove as much of the uneaten plastics as possible, this was to ensure each feed was the precise concentrations required for the treatment. Each replicate aquarium housed four fish, except for clutch B where only 32 individuals were available, and therefore, aquaria housed three or four individuals (Figure 5.1). All aquaria contained short sections of PVC pipe at the base of the aquarium as shelters for the fish and mesh over the outflow (and aquaria tops) to prevent fish loss. Individual fish were tagged using elastomer tags to enable measurement of growth of each individual fish over the experimental period.

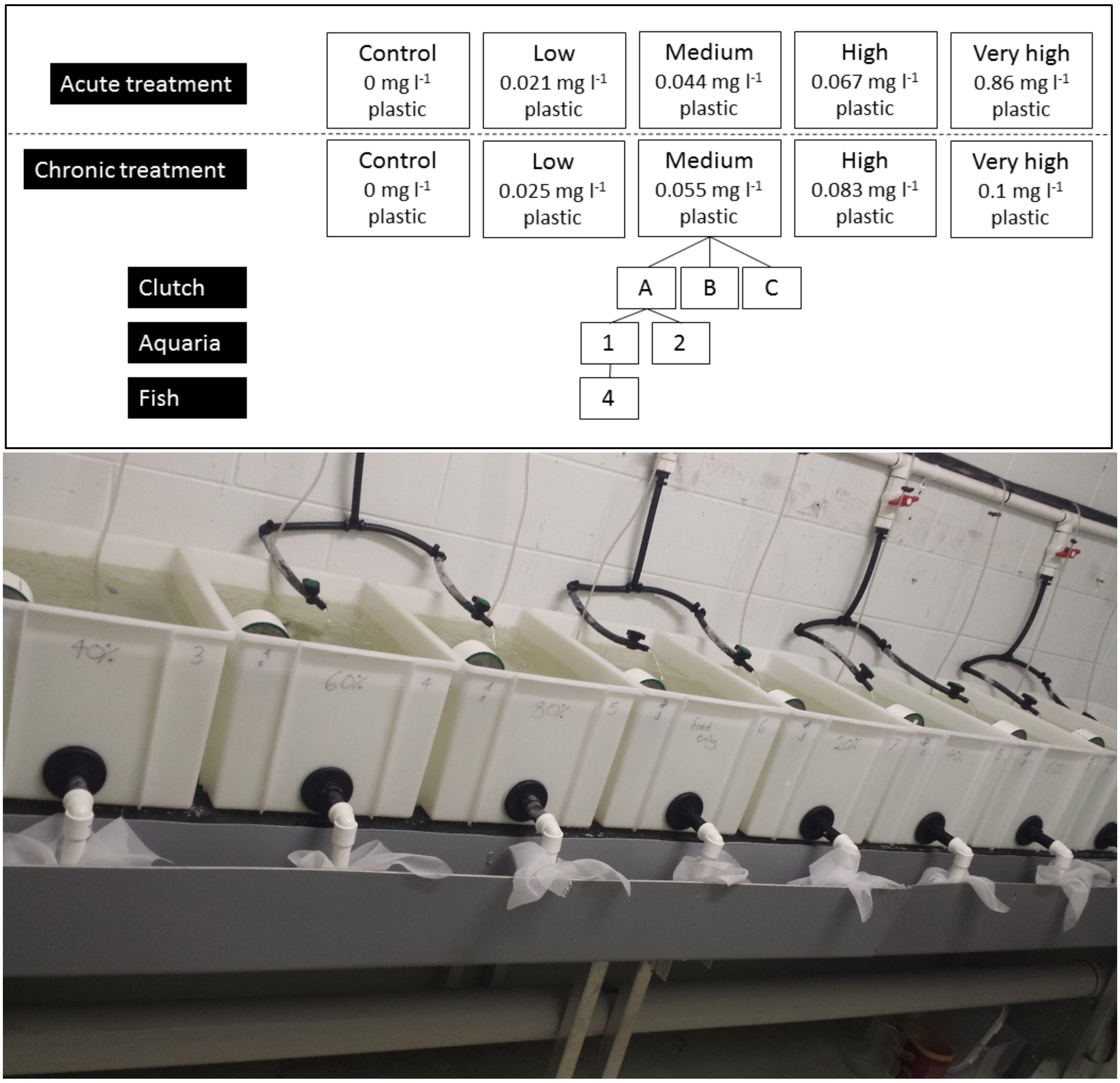


Figure 5.1: *Experimental design for acute and chronic plastic exposure experiment. Concentrations shown are the mean concentrations for each treatment, as treatment was dependent on tank biomass. The design of the aquarium room showing aquaria and outflow filters.*

5.2.3 Preparation of microplastics

Polyethylene terephthalate (PET) was used for the experiment as it is one of the most common plastics types found in the environment (Morét-Ferguson et al., 2010). For consistency with the food supplied to the fish during rearing, and the size of their natural prey, PET microplastics with a particle size approximately the same as the commercial food pellets were created by cryomilling post-consumer recycled plastic pellets, donated from Visy Plastics (Smithfield, NSW, Australia). The cryomilling process created a range of particle sizes, therefore the desired plastic size classes were obtained by sieving the cryomilled plastics with stacked sieves of the desired sizes. The commercial food pellets were approximately 2 mm in diameter and the plastics ranged from 1 to 2 mm. The plastic particles were divided into mesh bags and placed in indoor sump aquaria of a mature salt water aquarium system for at least two weeks prior to the start of the experiment. This was to encourage the growth of a microbial biofilm on the surface of the milled microplastics to better mimic microplastics in nature (Savoca et al., 2017). PET has a specific gravity of 1.38 and is, therefore, negatively buoyant in seawater. However, the small milled particles have a high surface area to volume ratio and remained at the water surface for ~2 minutes after being dropped into the water, behaving similarly to the commercial food pellets.

5.2.4 Acute exposure experiment – food replaced by plastic

During the first week of the exposure experiment, I assessed the impact that replacement of natural food products with plastic particles might have on fish. This 'acute' exposure was designed to indicate how fish growth might be affected in marine habitats where natural plankton concentrations are declining (evidence in (Boyce et al., 2010)) and microplastic concentrations are increasing (Ivar do Sul and Costa, 2014). During this week, the fish (N=112) received a total 'food' allowance between 0.022 g to 0.065 g (0.055-0.16 mg l^{-1}) per feed, depending on aquaria fish biomass, which included different proportions of food and PET (see Appendix 3, Table 1).

5.2.5 Chronic exposure experiment – plastic dose added to normal food

After the 'acute' exposure phase, fish were exposed to plastic concentrations which consisted of their normal food (biomass adjusted) plus either zero (control group), low (20%), medium (40%), high (60%) or very high (80%) percentage of their diet added as plastic (see Appendix 3, Table 1). No tank received more than 0.065g (0.16 mg l^{-1}) food per feeding bout, as excess food in the system can reduce water quality.

5.2.6 Ontogenetic changes in sizes of microplastics ingested

A second group of fish (n = 69) were sourced from stock *A. polyacanthus* at MARF and were divided into two size classes, based on fork length, with small fish 30 to 35 mm and large fish 35 to 45 mm. Fish from these two treatment groups were randomly allocated among three different plastic particle size treatments (small, 125-300 µm [approx. 140000 per g]; medium, 300-1000 µm [approx. 5000 per g]; and large, 1000 – 5000 µm [approx. 60 per g]), with 3 replicate aquaria per treatment with three to four fish per aquarium. The fish were fed twice daily with a diet of commercial pellets (amount calculated based on fish biomass in each aquaria as above) with additional plastic 80% of the food mass (0.05 to 0.13 mg l^{-1} per feed, see Appendix 3, Table 2). I acknowledge that using weight of particles resulted in a different number of particles being supplied to each tank depending on particle size. The aim of using high concentrations of particles was an attempt to make consumption rate limited by particle processing time, rather than the number of particles available (a type II functional response) (Staddon, 2016). After one week of exposure, the fish were euthanized and dissected to assess the amount of plastic retained in the whole length of the GI tract of each animal.

5.2.7 Measured response variables

Plastic retention, growth and body condition

At the conclusion of the plastic exposure experiments, the fish were euthanized according to standard animal ethics protocols, by MS-222 overdose (AVMA Guidelines for Euthanasia of Animals, 2013). The gut content of the fish was collected to determine the amount of plastics the fish retained. This plastic ingestion quantifies the amount of plastic ingested and retained in the gut over both the acute and chronic phases of the exposure experiment. The same procedure was followed at the end of the seven day duration particle size experiment.

I collected weekly length and weight data to assess growth through the acute and chronic exposure phases. Each fish was photographed using a camera fixed a set distance from a gridded background, to accurately measure length and width using 'Image J' software (Figure 5.2A). The wet weight of each fish was measured by adding the fish to a beaker of aquarium water on pre-zeroed scales (Kern PBC 3-place analytical balance, Kern and Sohn KmbH, Germany Figure 5.2B). Individual growth rates were calculated, as change in mass and change in length, weekly and over the duration of the whole experiment. Length-weight ratios were calculated for each fish at the beginning and end of both acute exposure and chronic exposure phases. The change in this ratio gives an indication of the change to the body condition of each fish over each phase of the experiment. During the post-exposure dissections, the liver of each fish was extracted and weighed (ME235P, 5-place Sartorius analytical balance, Germany) to calculate the liver weight to body weight ratio (Hepatosomatic index; Equation 1), and used to assess the body condition of each fish (Chellappa et al., 1995):

$$\text{Hepatosomatic index (HSI)} = (LW/BW) * 100 \qquad \text{Equation 5.1}$$

where LW is liver weight and BW is the total body weight of the fish before dissection. The HSI reflects the energy reserves in the liver and, as liver function is critical for overall health, is a reliable proxy for condition (Pereira et al., 1993).

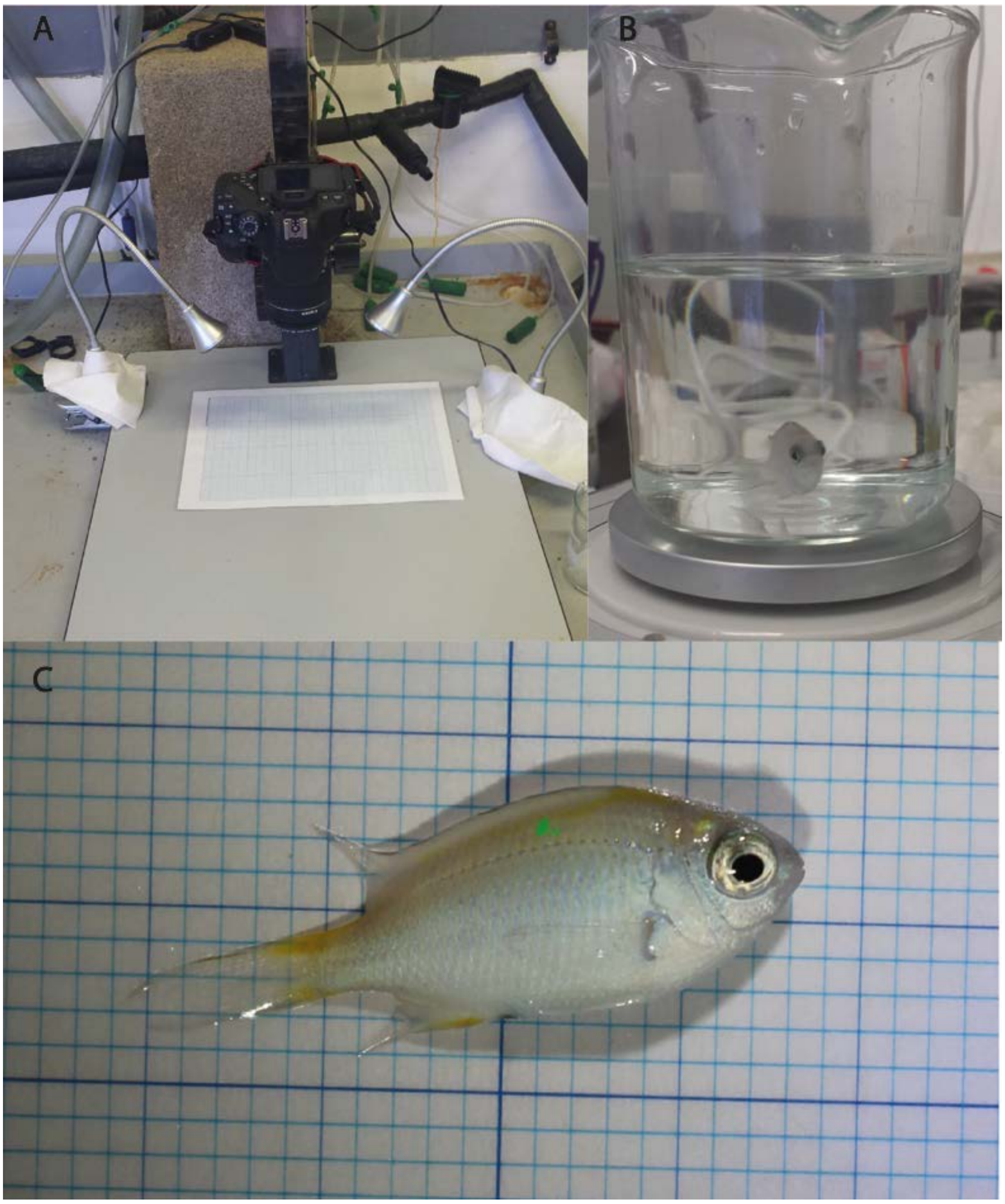


Figure 5.2: *Set up of length (A) and weight (B) data collection. A) shows the camera, grid paper and lighting set up for taking images of the fish to length analysis, and B) shows a juvenile* A. polyacanthus *in a two litre beaker of aquarium water on a balance. C) shows an example of a photograph used to measure the length, the green elastomer tag can be seen.*

Behaviour

Effects of microplastic exposure on fish behaviour were assessed using two sets of video observations at the end of the chronic exposure phase. These video observations monitored: 1) fish swimming activity and aggression between feeding times (20 minute videos, taken in between feeding times between the hours of 10 am and 4 pm); and 2) foraging behaviour and aggression at feeding times, for two minutes after food was introduced to each aquarium. To ensure the fish were behaving "normally" after the disruption of camera placement, the camera was set up and set to record 10 minutes before the introduction of food. I used a GoPro (Hero 3) camera that was affixed to a stand that kept the camera at a set distance from the bottom of the aquaria.

Fish swimming activity was measured between feeding times, using the 'grid crossings' method (e.g. Ryer et al., 2009). Briefly, a 2 x 2 cm grid (corresponding to approximately one body length of *A. polyacanthus* in our videos) was placed over the computer screen during video playback, and the number of times each individual fish crossed a grid line was counted during a period of 2 minutes. The fish was considered to have crossed a gridline if the nose and both eyes crossed the line. Data are reported as line crossings per second. For this metric I used the longer inter-feeding videos with the 2 min sample taken at least 10 min after the start of the video to ensure the behaviour had returned to "normal" after camera set up.

Aggressive behaviour between individual fish was assessed from the videos taken during and after feeding, and was measured as active interactions between individuals where a dominant fish caused a subordinate to move position within the aquaria. The total number of aggressive interactions in the focal aquaria were counted during the remainder of the video, which ranged from 5 - 10 minutes (depending on video length) after the food was introduced. I used the maximum video time available to gain the maximum data, the count was then made relative to time in the aggression index described below (Equation 2). The aggressive interactions observed in each aquarium were categorised into short and long interactions. The short interactions (hereafter called 'swoops') were counted as lunges or movements of a dominant fish that caused a subordinate fish to move away, whereas long interactions (hereafter called 'chases') were counted as interactions where the dominant fish chased a subordinate fish for approximately ¼ the aquaria length (assessed visually; not measured for every interaction). An aggression index (AI) was calculated to account for the different energy costs of swoops compared with chases, due to the different levels of swimming activity and different duration of activity, and was normalised by the duration of the observation period as:

$$AI = (S + 2C)/t \qquad \text{Equation 5.2}$$

where S is the observed number of swoops, C is the observed number of chases, and t is the duration of the observation period (decimal minutes). This creates a measure of aggression experienced by the fish per aquaria. All the aquaria were the same dimensions and colour to ensure there was no confounding effect of environment on the treatment groups.

5.2.8 Data analysis

Analysis of the length-weight relationship of the experimental fish showed that three different clutches started with different sizes, and different weights per unit body length (See Appendix 3 Figure 1, ANOVA, $F = 747.9$, $df = 5, 106$, p-value < 0.001). Consequently, 'clutch' was retained as a factor in subsequent analyses.

To assess differences in plastic ingestion between treatments from the chronic exposure experiment, I used a generalized linear mixed effect model, fit by Laplace Approximation maximum likelihood in the lme4 package in R (Bates et al., 2015; R Core Team), with the Kenward-Roger approximation to obtain p-values. To understand if the number of plastics retained during the chronic exposure varied between treatments, I fit a linear mixed effect model to the plastic consumption data for the subset of fish that ate plastics, again using the lme4 package. Due to the small number of fish that consumed plastics, there was insufficient statistical power to analyse these data among individual treatments and, therefore, the treatments were pooled into high (60% and 80%) and low (20% and 40%) treatments for this particular analysis. Consumption levels by fish of different sizes, for each of the plastic sizes were assessed using the Fisher's exact test using a multi-level contingency table analysis.

Generalized linear mixed effect models were fitted to growth, body condition, hepatosomatic index, and line-crossing data, to assess effect of treatment and clutch on fish health, with aquaria included as a random effect. It was important to include aquaria as a random effect to remove the artificial inflation of statistical replicates, while maintaining the individual fish data. These analyses were conducted using the nlme package for R (Pinheiro et al., 2018). Linear models using the nlme package were also fit to the length and weight data to quantify the relationships between and within clutches, and the difference in the slope/intercept of the relationship was assessed to understand body condition during the acute phase of the exposure experiment. To test the effect of plastics exposure on the activity of the fish, the line-crossings per second were compared using a Levene's Test for homogeneity of variance. An ANCOVA was conducted to test the relationship between treatment and aggression with the aggression index data.

5.3 Results

5.3.1 Plastic ingestion

Chronic exposure experiment

At the end of the chronic phase of the exposure experiment, 19.6% of fish had plastic fragments in their GI tracts. For these fish, the number of retained plastic fragments ranged from one to eight (Figure 5.2) and there was a general trend toward higher ingestion at intermediate plastic exposure concentrations (Figure 5.2). However, neither the proportion of fish that had retained plastics (Figure 5.2A-C), nor the number of plastics retained, were dependent on the concentration of plastic present in aquaria (Figure 5.2D-F), and these responses did not vary between clutches (Table 5.1). Due to an error by a volunteer, on one occasion control tank 11A was fed a 20% dose treatment, which resulted in one control fish having one piece of plastic in their GI tract (Figure 5.3A).

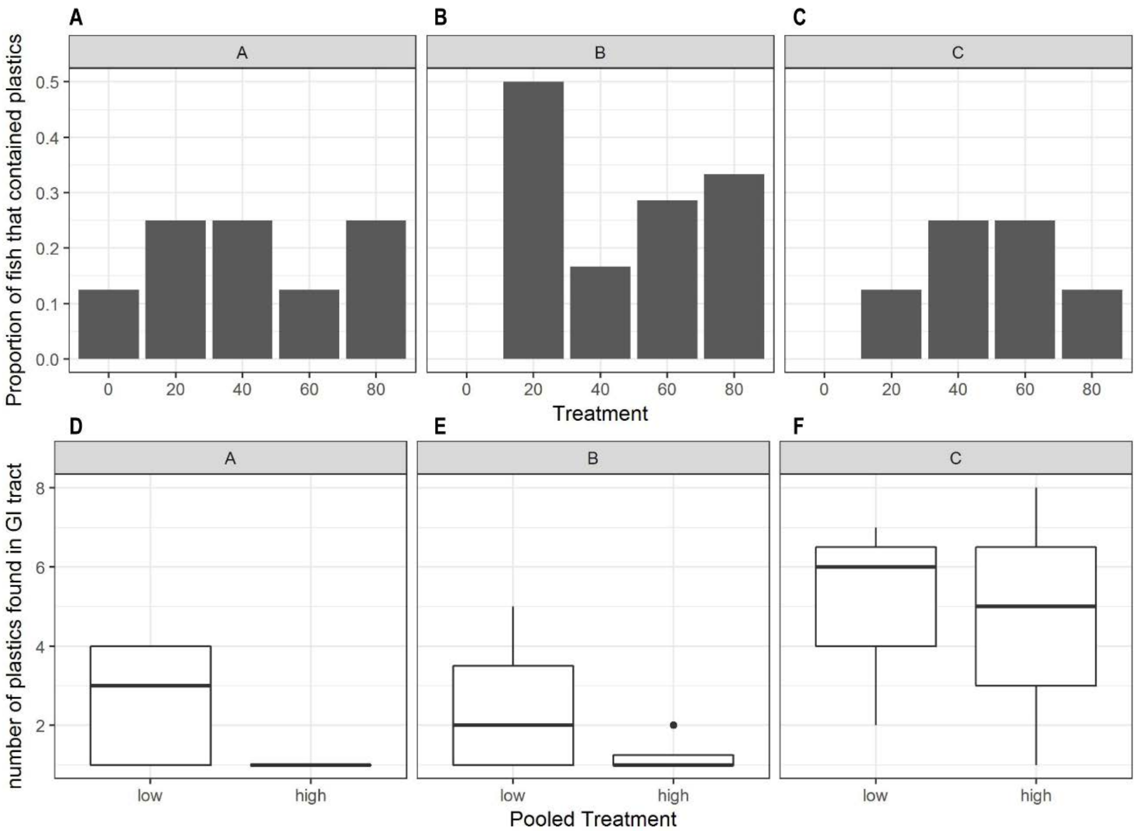


Figure 5.3: *Plastic ingestion by sub-adult A. polyacanthus exposed to different plastic doses. Top panels (A-C) show proportion of fish per treatment and clutch which ingested plastics, and the lower panels (D-F) show the range of ingestion for fish per pooled treatment (treatments 20, 40 pooled to low, and 60, 80 pooled to high) for each clutch, denoted in the grey panel above each graph.*

Ingestion of different particle sizes

The number of retained plastics was strongly dependent upon plastic particle size. All fish exposed to the smallest plastic fragments were found to have plastics in their GI tract after one week of exposure (Figure 5.3), and over half of the fish exposed to medium sized plastics were also found to have plastics in the GI tract (Figure 5.3). The observed proportions of fish of different sizes that had consumed small and large plastics was significantly different than random (Fishers Exact Test, p-value < 0.001). Larger fish appeared to be more likely to ingest plastics of all sizes than smaller fish (Figure 5.3A). For each particle size treatment there was no statistical difference in amount of plastics consumed between the fish sizes (Figure 5.3B). However, there was a large difference in the amount of plastic consumed between particle size treatments, with fish ingesting up to a maximum of 2102 small plastic fragments (in a fish classed as large) in comparison to a maximum of 5 in the large plastic particle treatment (Figure 5.3). Fish were found to have much higher numbers of plastics in their GI tract when exposed to the small particle treatment (Figure 5.3B, $F = 15.523$, $df = 12$, p-value $= 0.0005$).

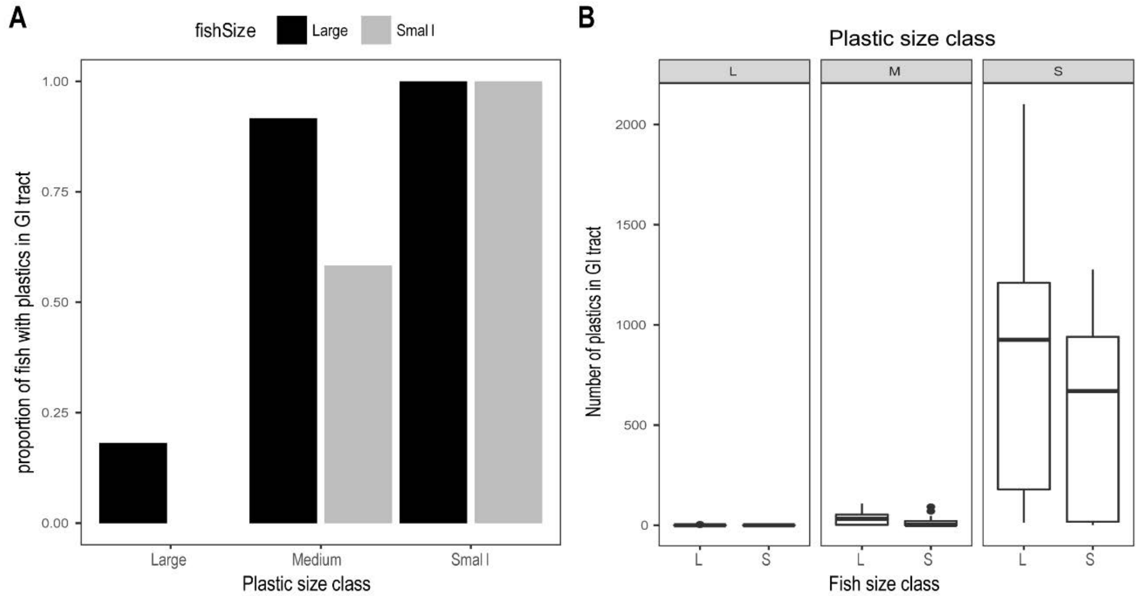


Figure 5.4: *Particle and fish size dependence of plastic ingestion. A) Shows the proportion of fish that were found to have plastics in their digestive tract. B) Shows the range of plastic ingestion for fish per treatment, denoted in the grey panel above each graph. The mid-line of the boxes represent means with the boxes showing the 25th and 75th percentile and the vertical lines representing the range.*

5.3.2 Growth and Body Condition

Growth over the different exposure regimes

Fish growth during the acute exposure phase of the experiment was the lowest in the higher plastic treatments (Figure 5.4); these fish were receiving much less food than the control fish. However, the different clutches also reacted significantly differently in this experiment (Table 5.1). Clutch A and C had negative growth in the highest plastic treatment while clutch B had reduced, but not negative growth. Conversely, the chronic exposure phase showed very little effect of plastic presence (Figure 5.5). During this phase, the fish that had lost mass during the acute phase seemed to catch up with the rest of their cohort and by the end of the chronic exposure phase (6 weeks) there was no significant difference between treatments in the relative change in body mass. There was a significant difference in the way clutch C responded compared to the other clutches (Table 5.1), but within clutch, there was no significant effect of plastic concentration.

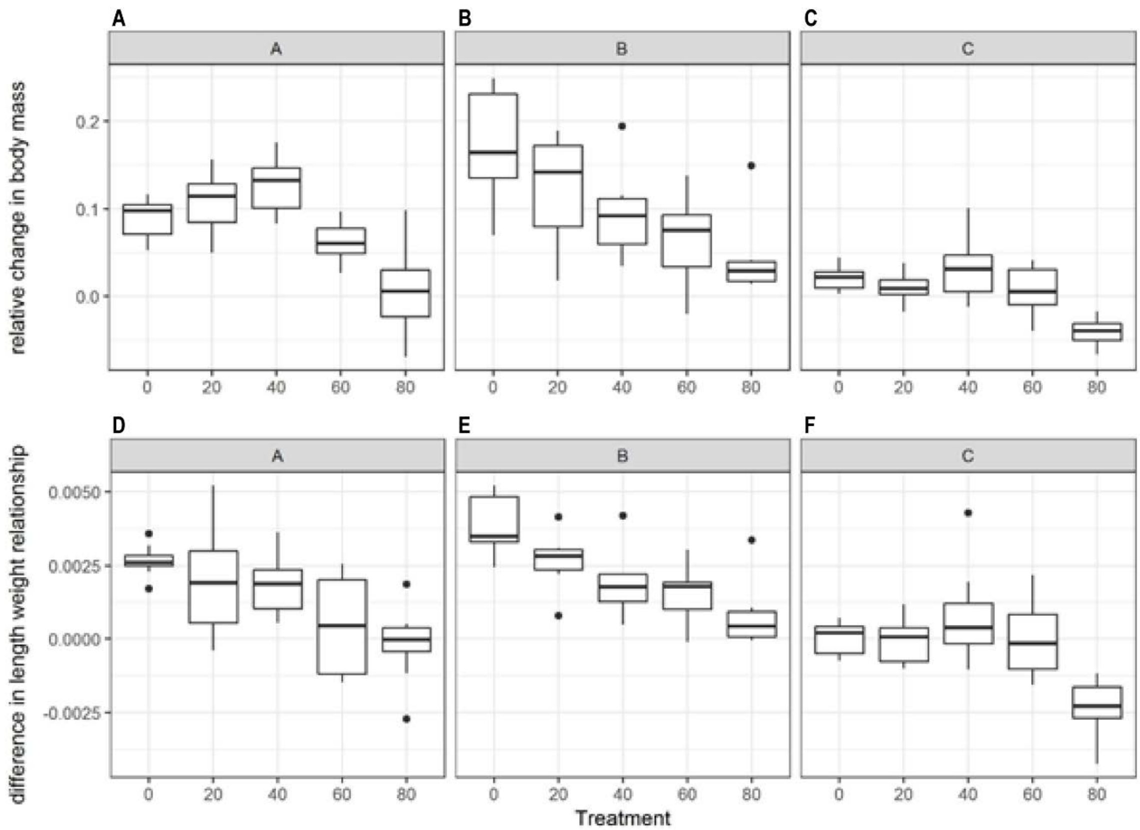


Figure 5.5: *Fish growth and body condition during the acute phase of the plastic exposure experiment. Top panels (A-C) show the change in mass (g) relative to start mass during the acute exposure phase, for each clutch. The lower panels (D-F) show the change in length-weight relationship relative to the start of the acute exposure, for each clutch, denoted in the grey panel above each graph. The mid-line of the boxes represent means with the boxes showing the 25th and 75th percentile and the vertical lines representing the range.*

Body condition

Based on the difference in length-weight relationship between the start of the chronic exposure and the end of the experiment, there was no significant change in body condition due to the treatments (Table 5.1). The only exception to this general pattern was observed in clutch A, where there was an increase in body condition in the 40% plastic treatment compared with the control. Based on the hepatosomatic index, there was a general decrease in body condition in higher plastic concentration treatments (Figure 5.5D-F). This decrease was most pronounced for clutch C, for which condition was lower in the 80% treatment, however, this trend was not consistent across clutches.

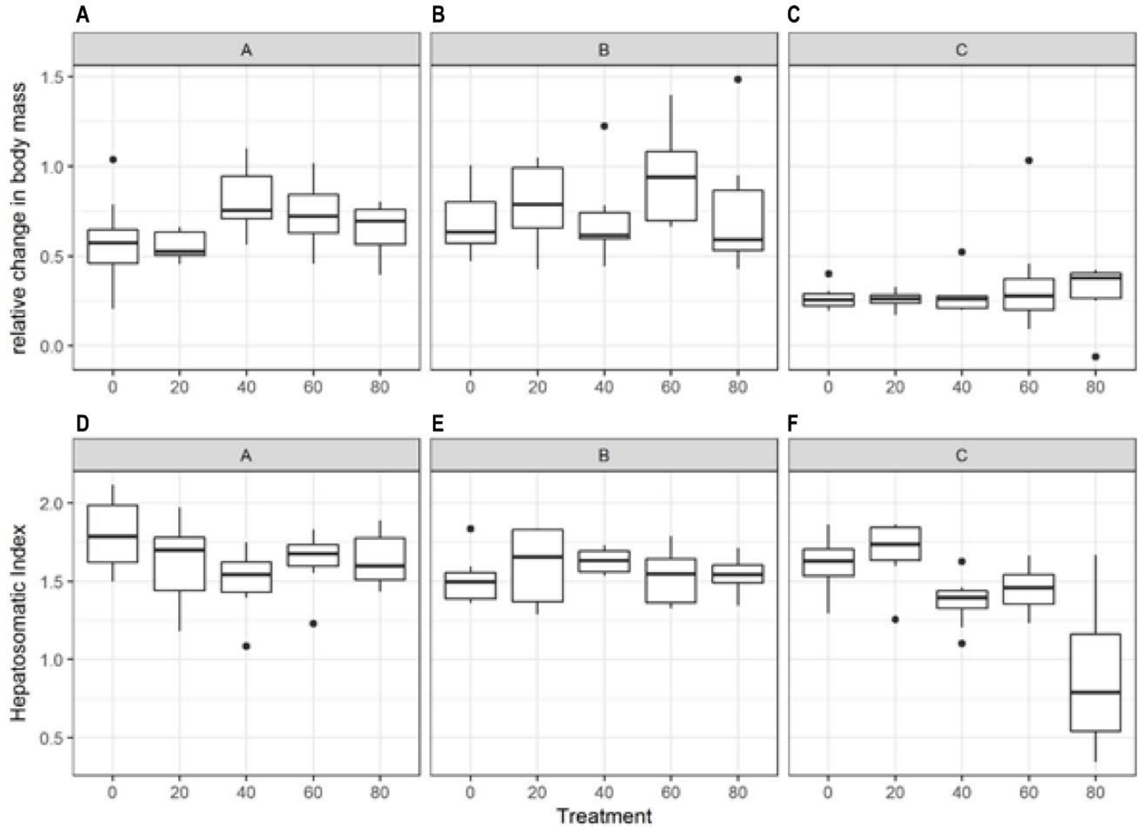


Figure 5.6: *Fish growth and body condition (measured as HSI) during the chronic exposure phase of the experiment. The top panels (A-C) show the change in body mass relative to the start of the chronic exposure, for each clutch, denoted in the grey panel above each graph. The lower panels (D-F) show the body condition of the fish in the form of the Hepatosomatic Index in each treatment and clutch, denoted in the grey panel above each graph. The mid-line of the boxes represent means with the boxes showing the 25th and 75th percentile and the vertical lines representing the range.*

5.3.3 Behaviour

Different clutches showed a different behavioural response to the plastic exposure treatments (Figure 5.6, Table 5.1). For clutch C, there was a general increase in activity (measured as line crossings) with increasing plastic concentration, but the same trend was not apparent for the other two clutches (Figure 5.6A). Overall, there was generally high variability in activity among individuals. There was a significant association between the amount of aggression and the number of line crossings observed in each tank (Pearson's Test, r = 0.8097, df = 7, p=0.0082). Indeed, the majority of the more aggressive interactions (chases) resulted in a long distance travelled for both the aggressor and the subordinate fish leading to an association between these behavioural metrics. Similar to the observed variation in line crossings, the intensity of aggression was highly variable between tanks within treatments groups. Although aggression was higher, on average, for clutch A (AI of 9.7 compared with 7.5 and 7.2 in clutch B and C, respectively, Figure 5.6B), ANCOVA with Tukey's post-hoc test showed that these differences were not statistically significant. There was also some indication for clutch B of a decline in aggression with plastic concentration, but this trend was not statistically significant (Figure 5.6B).

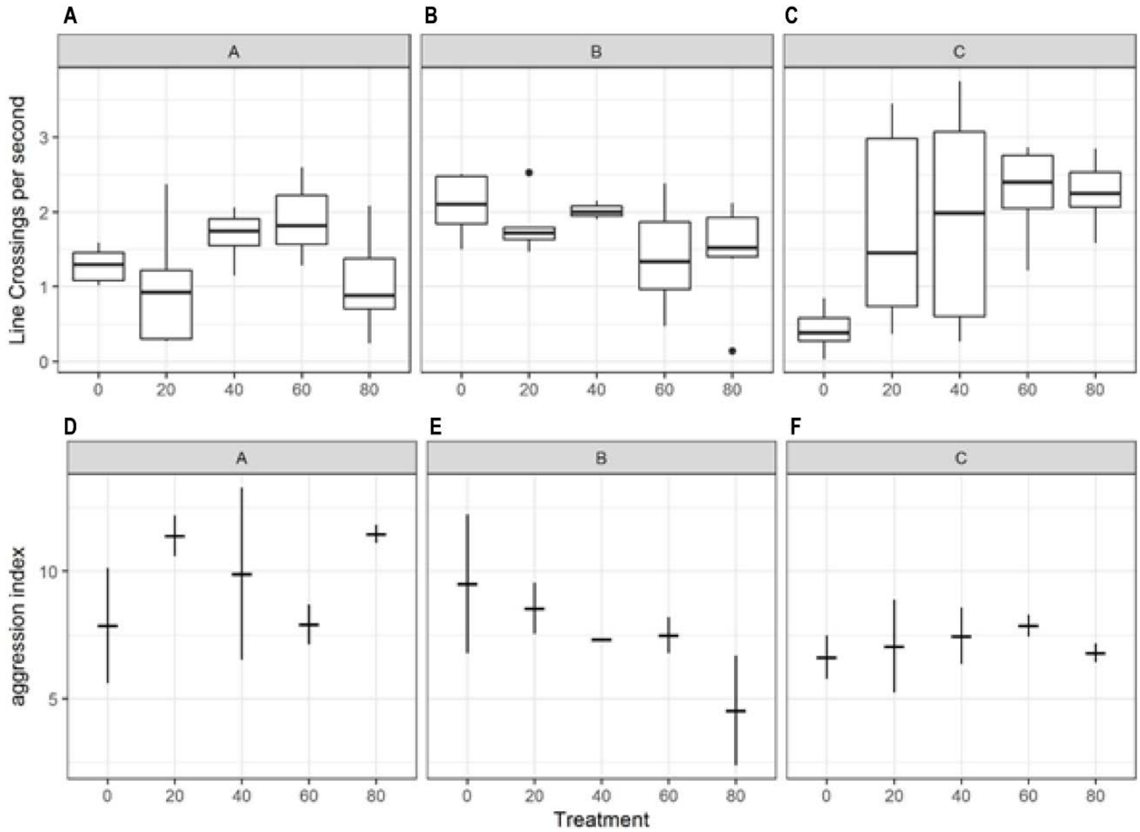


Figure 5.7: *Fish behaviour during exposure to different concentrations of microplastics.* *The top panels (A-C) show the number of lines crossed per second for each treatment and clutch, denoted in the grey panel above each graph. The mid-line of the boxes represent means with the boxes showing the 25th and 75th percentile and the vertical lines representing the range. The lower panels (D-F) show the aggression index where the central horizontal line shows the mean and the vertical line indicates the range of values.*

Table 5.1: Summary table of statistical analyses.

Analysis	Factor	Df	Test statistic	p	Figure
Proportion of fish that ingested plastic	Treatment	4,96	*F* = 0.001	0.900	2A-C
	Clutch	2,96	*F* = 0.006	0.900	
	Treatment by Clutch	8,96	*F* = 0.02	0.900	
Number of plastic particles eaten by fish	Intercept	1,11	*F* = 42	<0.001	2D-F
	Treatment (pooled)	1,11	*F* = 1.2	0.290	
	Clutch	2,11	*F* = 4.9	<0.050	
	Treatment by Clutch	2,11	*F* = 0.22	0.800	
Size-dependence of plastic ingestion	Intercept	1,51	*F* = 17	<0.001	3
	Treatment (pooled)	1,12	*F* = 0.70	0.420	
	Clutch	2,12	*F* = 16	<0.001	
	Treatment by Clutch	2,12	*F* = 0.40	0.680	
Growth – acute phase	Intercept	1,82	*F* = 151.4	<0.001	4A-C
	Treatment	4,15	*F*= 11.1	0.002	
	Clutch	2,15	*F*= 34.1	<0.001	
	Treatment by Clutch	8,15	*F=1.66*	0.189	
Body condition – acute phase	Intercept	1,82	*F* = 33.7	<0.001	4D-F
	Treatment	4,15	*F*= 6.75	0.003	
	Clutch	2,15	*F*= 16.01	<0.001	
	Treatment by Clutch	8,15	*F*= 0.74	0.655	
Growth – chronic phase	Intercept	1,82	*F* = 554.6	<0.001	5A-C
	Treatment	4,15	*F*= 1.36	0.294	
	Clutch	2,15	*F*= 36.5	<0.001	
	Treatment by Clutch	8,15	*F*= 0.58	0.776	
Body condition – chronic phase	Intercept	1,82	*F* = 786.8	<0.001	
	Treatment	4,15	*F*= 2.53	0.084	
	Clutch	2,15	*F*= 6.47	<0.001	
	Treatment by Clutch	8,15	*F*= 0.91	0.532	
Hepatosomatic index	Intercept	1,82	*F* = 2267.8	<0.001	5D-F
	Treatment	4,15	*F*= 2.66	0.074	
	Clutch	2,15	*F= 4.56*	0.028	
	Treatment by Clutch	8,15	*F= 2.17*	0.093	

5.4 Discussion

In this study, I found that when microplastic particles were the same size as a fish's normal food, rates of plastic retention in the gut were generally low, and were independent of the concentration of microplastic available in seawater. Moreover, this low-level plastic consumption had negligible effects on the growth and body condition of juvenile planktivorous reef fish after six weeks of exposure. However, when plastic particles replaced food there was a significant decrease in growth and body condition compared with controls due to limited food availability. Our results also indicated that plastic consumption and/or plastic presence in seawater did not affect fish foraging behaviour (activity) or aggressive interactions. However, the amount of plastic found in the GI tract was influenced by microplastic particle size, with smaller microplastics much more frequently consumed than larger microplastics, regardless of fish size. These results support the hypothesis that reduced plastics size will increase the rate of consumption as the ability to distinguish between food and non-food particles decreases.

The low rates of plastic particle retention in the GI tract (up to eight particles per fish) for microplastics 1 – 2 mm diameter observed in this study suggests that juvenile *A. polyacanthus* can recognise and avoid consuming microplastic particles of certain sizes, or can readily eject plastic particles after consumption. The number of particles in the GI tract of the fish found here is generally consistent with field studies. For example, Lusher et al., (2013) found small quantities of microplastic particles (one to 15 per fish) were commonly ingested by fish in the natural environment regardless of fish species and feeding habitat. A similar range of values was found by Neves et al., (2015) and Foekema et al., (2013), who found only one to four particles per fish, with most fish containing only 1 particle. Field-sampled fish tended to have mostly plastic fibres in their GI tract, as opposed to particles as assessed in this study, suggesting that fish can more readily recognise and avoid certain types and/or shapes of microplastics. Hermsen et al., (2017) suggest many of these fibres could be a result of contamination from sample preparation methods, and this should be considered in future studies. Nevertheless, in my study the particulate microplastics used in the experiments were easily distinguishable from fibres and I found that, when plastic particles were less than one quarter the size of normal food (300 - 125 µm), retention rates were vastly higher. Collectively, these results indicate that when fish are exposed to a variety of sizes of plastics, smaller plastics will be ingested more readily than larger ones. This dramatic variation in the number of plastics found in the GI tract of different size classes of plastics suggests that as plastics get smaller they may be less readily differentiated from normal prey (O'Brien et al., 1976; Foekema et al., 2013; Eriksen et al., 2014). While I did not quantify the effects of ingestion of these small microplastics, it is likely that very small plastics cause less damage and/or blockage to the digestive system. There is a possibility that fish could reach satiation based on volume not number of particles, therefore, consumption of a many very small particles could have a less impact to the individual, than fewer larger ones. However, the increased surface area to volume ratio of smaller microplastics increases the possibility that associated contaminants or toxins could leach from the microplastics and be absorbed by the fish. Further studies are required to assess whether microplastics will have a larger effect on fish populations as they break up into smaller pieces, as they are more readily consumed and have higher potential to transfer contaminants (Bakir et al., 2012).

Under acute exposure conditions, where microplastic particles replaced food particles, the fish grew slower and lost body condition, and these effects were larger at higher microplastic concentrations. The effects of a reduction in food availability on fish growth and survival are well understood (e.g. Elliott, 1973). Indeed, the minimum food requirements for growth of various fish species have been quantified in aquaculture and natural settings (Elliott, 1976; Garvey et al., 2004; Imre et al., 2004). Consequently, changes in fish growth and condition during this phase of the experiment likely reflect the reduction in food ration rather than any toxic effects of microplastic exposure. Although I did not measure plastic retention in the gut at the end of the acute phase, plastic retention was low at the chronic phase of the experiment. These results indicate that plastic ingestion is low even when food availability is restricted, and/or that small PET fragments plastics pass through the digestive tract and are evacuated with minimal harm to fish. My results also show that fish with decreased, and even negative, growth under low food (high microplastic) treatments during the acute phase were the same size, on average, as fish in the high food (low microplastic) treatments by the end of the chronic phase. Such, 'compensatory' growth is commonly observed in fish (Hurst et al., 2005). In my study, fish 'caught up' in size within 6 weeks, highlighting that sub-adult fish of this species favour growth at the early stage of life to reduce predation risk for the individual (Metcalfe and Monaghan, 2001). The literature documents that plankton and plastics are both patchy in nature (e.g. Bengfort et al., 2014). Consequently, fish might face intermittent periods of low food rations that have negative effects on fitness by slowing growth rates, and by flow-on effects associated with compensatory growth of juvenile and sub-adult individuals.

Due to the low retention rates and non-significant effect of microplastic exposure (1 – 2 mm), the consumption of one microplastic in an individual that was in the control treatment was unfortunate, but not significant to the overall conclusion of this chapter.

5.4.1 Behaviour

The behavioural measures used in this study (activity and aggression) were directly correlated, suggesting that in aquaria that had more active fish, the fish were being active because of the aggressive interactions taking place. The dominant and most aggressive fish often utilised the shelter, and would chase subordinate fish away from the entrance (KC personal observation). Low (1971) reports subordinate fish being driven a total of five metres in a five minute period of field observations of reef fish (Family *Pomacentridae*). Dominance and space use are tightly correlated, including feeding position (Hughes, 1992) and shelter use (Phillips and Swears, 1979). The feeding dynamics of a group can also change with changing dominance hierarchy (e.g. Forrester, 1991; David et al., 2007). I found no significant relationship between activity and treatment, with large amounts of variation between and within clutches. There is a slight suggestion that fish are more active at medium levels of exposure. This would support findings of increased aggression at medium feed levels by Toobaie and Grant (2013). It would have been valuable to my study to have behavioural observations before and after exposure, as this would have allowed direct observation of changes to behaviour. I observed vastly different feeding behaviour between individuals and aquaria, suggesting some individuals or groups within a population may be more affected by plastic consumption than others. Indeed, Sparks et al., (1972) suggest that a dominance hierarchy can greatly affect the outcome of ecological experiments.

The rate of interactions between individuals and plastics may influence the rate of learning of the individual. There is strong evidence that individual fish learn quickly to avoid anglers in catch and release fisheries, which have strong learning experiences (Askey et al., 2006). However, when the plastic particle size is small, I suggest the fish do not have the same learning experience, as the plastics have the potential to simply pass through their GI tract with little to no discomfort. This allows plastics to enter the GI tract, potentially transferring plastics-associated toxins, possibly causing harm to the fish, without an avoidance behaviour to stop it. In most parts of the oceans, fish would currently encounter plastics very rarely, not allowing the learning experience (e.g. < 1 MP m^{-3} in East China Sea; Zhao et al., 2014). In areas with high microplastics load, for example the North-eastern Pacific (~280 MP m^{-3}; Desforges et al., 2014), the fish may actually be more equipped to avoid them. The size and shape of particles that could be avoided are probably based on fish species, feeding type, gape, etc. that affect the learning experience of the individual fish. When particles become so small that they are imperceptible to the fish, it is likely they will have different impacts to the fish, including gill irritation (Kashiwada, 2006).

To summarise, I found that juvenile *A. polyacanthus* can be tolerant of plastic exposure, finding no significant effect on growth, body condition or behaviour while the plastic particles were the same size as their natural food particles. However, plastic retention rates vastly increased when the size of the plastic particles was reduced from ~2 mm to 300 - 125 µm. This is of concern because plastics in the environment fragment into smaller and smaller pieces and, therefore, could become more readily ingested by planktivorous fish.

Chapter 6
General Discussion

Plastic pollution in the environment has been shown to have dramatic negative effects on habitats, wildlife, local economies and environmental values (Derraik, 2002; Thompson et al., 2009; Jang et al., 2014; Krelling et al., 2017). There have been many studies documenting the observed negative interactions of marine species with plastic pollution (see reviews by Thompson et al., (2009), Andrady (2011), and Chae and An (2017)). Plastic pollution can also cause damage to marine habitats through smothering, scouring (e.g. Uneputty and Evans, 1997; Donohue et al., 2001) and changed physical properties (Carson et al., 2011). Plastic pollution accumulation is negatively impacting local economies (Jang et al., 2014; Krelling et al., 2017). Consequently, plastic pollution has become a globally ubiquitous issue for the marine environment, and there is clearly a pressing need for management action at multiple spatial scales. However, although it is a global issue, management policies and actions are generally implemented at the local municipality scale, and local management authorities are usually the entity with the power to reduce the input from local sources.

The management of plastic pollution is challenging because the agencies or levels of government responsible generally have small spatial jurisdictions – e.g. municipal areas. At the scale of a local management authority, mitigating plastic pollution requires either cleaning up existing debris or reducing inputs, or preferably both. These actions require data on where plastic pollution is coming from and where it is going within their management area, because understanding this transport can identify sources, which is crucial for targeted management action. The problem for local managers usually occurs because they generally don't have the data they need at the resolution or scale that is useful to them, and are often under-resourced in capacity or finances to collect that data. The available data describing plastic transport and sources is often at larger spatial scales and the resolution is too coarse for a small management area (Johnson and Eiler, 1999; Hinata et al., 2017).

The overarching goal of this thesis was to understand the dispersal and risks of plastic pollution at a management-relevant scale. Collectively, the results of my thesis show there is large variability in the dispersal and accumulation of plastic pollution in the coastal environment, and the effects of that accumulation are also highly variable. The thesis aims can be split into those pertaining to dispersal and those pertaining to risk. In this chapter I will discuss each of the aims of this thesis in the context of existing research and provide a synthesis of findings. I will also discuss the approach used in this thesis, the implications of my research, and outline opportunities for future research.

6.1 Dispersal and accumulation of plastic pollution in the coastal zone

One of the problems of understanding sources and accumulation of plastics in the coastal zone is the lack of understanding of dispersal processes. The first aim of my thesis was to understand the dispersal of plastics in the coastal zone at the scale of local management jurisdiction. The results I present in this thesis provide an empirical foundation for modelling plastics dispersal at small, management-relevant scales, especially in topographically complex regions, such as Queensland, Australia.

There are many processes driving the dispersal and accumulation of plastics in the coastal zone. In Chapter 2 I assessed the sensitivity of a plastic specific hydrodynamic model to: the rate of resuspension of plastics from the coastline; degradation of macroplastics into microplastics; the wind shadow effect; the wind-drift coefficient; settling rate; and the diffusivity parameter. I show that the variability in the model predictions is large when model parameters are adjusted, which suggests each process can influence the distribution of plastics in the coastal zone and highlights the need to obtain accurate estimates for these parameters in natural systems. In Chapter 4 I further highlight the need to include plastic-specific processes in dispersal estimates of plastic pollution. These many plastic-specific dispersal processes caused large variability in the field data presented in Chapter 4. Indeed, the accumulation I observed on the beach is the result of the supply and the removal of plastics from the beach, and the observed quantity is heavily dependent on the residence time of plastics on the beach. Hence, the supply and removal rates of plastics in the environment are influenced by a variety of processes. This is partially demonstrated in Chapter 2 where I show source to be the most influential model parameter dictating plastic movements and, in Chapter 4, the results indicate that macroplastics found on beaches are more likely to be from local compared with external sources.

In my thesis I explored accumulation of plastics on local beaches through the supply rate. Export is taken into account in the model in the form of resuspension of simulated macro- and microplastics from the coastline and degradation of simulated macroplastics into microplastics. However, the degradation parameter I applied in the model was different for beached and suspended simulated plastics, but was spatially uniform for all beached particles, and all particles at sea. Given that the degradation parameter of the model had a moderate impact on model predictions (Chapter 2), future work establishing estimates of degradation in a field setting should be prioritised (Weinstein et al., 2016; Welden and Cowie, 2017). Other factors influencing the export, mixing and degradation of plastic particles in the environment, not currently incorporated into the model, are: ocean properties (e.g. wave height and period of the waves), object properties (e.g. buoyancy and shape); and coastline (e.g. slope and nearshore bathymetry) or substrate properties (e.g. habitat and sediment type). Of these, wind-driven waves are likely to be particularly important. It is a limitation of SLIM that no wave field is included in the physics code. Many of these factors cannot be taken into account within the modelling environment directly at this stage, and may have contributed to the divergence between model predictions and data in Chapter 4.

The resuspension of plastics from coastlines is poorly understood. In Chapter 2 I explore the sensitivity of my model results to the method of resuspension from the coastline. From my results I conclude that the rate of resuspension is less important than whether resuspension occurs, concluding that beached particles must be allowed to re-suspend in future modelling studies. Resuspension is a large component of residence time but aside from estimates for large buoyant objects like fishing floats from tag/recapture type experiments (Johnson and Eiler, 1999; Hinata et al., 2017), or modelled estimates based on sequential clean-ups (Smith and Markic, 2013), there are few data to describe it. For smaller objects, Hinata et al., (2017) estimates the residence times of microplastics to be an order of four times lower than that of macroplastics. However, their estimate does not consider the dramatic reduction in upward velocity due to the reduced size of smaller particles. Although the assumptions of Hinata et al., (2017) are likely to be inaccurate, their estimates are a useful starting point for incorporating accurate resuspension parameters into future modelling.

Another factor missing from many existing plastic dispersal models is the wind shadow created when wind speed is reduced on the lee side of elevated topographic features. In Chapter 2, I found that incorporating wind shadow in the modelling of a topographically complex coastal zone is imperative at a management-relevant scale. The study area of this thesis has islands of many heights, which creates a complex topographic environment. Other hydrodynamic models for plastic dispersal have not generally included the wind shadow behind islands because it is less relevant at larger scales or in models with coarser resolution than mine. For example, Lebreton and Borrero (2013) model the debris generated by the 2011 tsunami at the scale of the whole Pacific Ocean, at a resolution of 1/12° (~9.2 km at the equator). At this scale and resolution, the wind shadow I used in this thesis (2500 m) would be insignificant within one of their cells. However, when the scale of the model domain is reduced, adding a wind shadow effect is essential for providing robust model results (Chapter 2).

Plastic pollution is made up of many different objects, each having different physical properties and hydraulic behaviour. This is one of the many challenges to modelling the movements of plastic pollution. For example, in Chapter 4 I present data showing that the movement of macroplastics, which tend to be larger and buoyant objects, are driven by the wind far more than microplastics, which are smaller and tend not break the surface. A method of reducing the variability in the model and obtaining a better fit with field data would be to focus on key informant objects that are relatively common in the field data and easy to identify (e.g. glow sticks used in longline fisheries, or plastic bottles). Then using experiments and field data to obtain key attributes of their hydraulic behaviours as inputs for the model, plastic type specific models could be run. This has been done successfully in a few cases: Ebbesmeyer et al., (2007) modelled a cargo spill (tub toys); Kako et al., (2011) modelled bottle caps; and Ebbesmeyer et al., (2011) modelled crab pots. However, in all cases the resolution and spatial scale were not suitable for local management action. Conducting a similar study in the GBR could be feasible, due to the extensive dataset of the Australian Marine Debris Initiative[1], which categorised the data by object type and thus it could be explored to select common items, or items with the highest risk to species, such as small plastic fragments.

6.2 Risks of plastic pollution to the coastal zone

The second aim of this thesis was to understand the risk of plastic pollution in the coastal zone at a management-relevant scale to species and habitats. I have shown that risk is highly variable in space and time. I show that the risk to a small planktivorous reef fish from microplastics of approximately two millimetres is negligible. The exposure to the reef habitat can be high (Chapter 3), but the consequence to planktivorous reef fish in that habitat is very low (Chapter 5) resulting in negligible risk. However, the exposure and consequence are potentially increasing as plastics continue to degrade into smaller particles (Chapter 5). I have shown that marine habitats experience variable exposure to plastics across seasons, and with the exception of habitats on the coastline (e.g. mangrove habitats) there is little consistent exposure across seasons.

Risk assessments are a useful tool for management allowing people to use an objective framework to prioritise management action (Bottrill et al., 2008). Risk assessments are in two quantifiable parts: the likelihood of a hazard event occurring; and the consequence to the value should the hazard event occur. Mitigating the effects of marine plastic pollution is difficult in the environment, essentially because the composition, distribution and/or quantity of plastic pollution in any local marine environment varies both spatially and seasonally. Thus, the likelihood

[1] https://www.tangaroablue.org/database.html

of plastic interactions in the coastal marine environment is variable in space and intensity (Chapter 3). Similarly, the consequence of exposure also varies over time according to variation in habitat use or diet at different stages of the life cycle of the organism and the availability of plastic particles to them (Chapter 5). These sources of variability result in a complex management problem. Many plastic risk assessments focus on the risk posed by plastic on average (e.g. Wilcox et al., 2013; Schuyler et al., 2014), however, my research shows that this is inadequate to truly understand the risks posed by plastics at a small scale, due to inconsistency of plastic accumulation at that scale.

The exposure (likelihood of interaction) of plastic changes in space and time, due to degradation, variability and patchiness of accumulation. Chapter 3 demonstrates that the location of the highest exposure changes with seasons, whereas Chapter 5 suggests that the exposure to plastics can have a negative consequence and could change on the timescale of plastic degradation (years). The physical location of accumulation is also variable at the timescale of season and day (Ryan et al., 2014; Hu et al., 2016). Other factors influencing exposure at small spatial and temporal scales are tide and wind as both can change convergence zones at a scale of hundreds of meters (or less) (Hu et al., 2016). Exposure at these smaller scales is important to understand when looking at beach-scale impacts, which is relevant for local management.

The consequence at locations of highest exposure must be explored further, particularly for habitats (Chapter 3). Lack of consequence data, especially how consequence changes with concentration, makes risk difficult to quantify at a species' population level in the environment. Chapter 5 demonstrates that the concentration of plastics of a certain size class has little effect on the growth or body condition of a planktivorous reef fish species posing a negligible risk of plastics of that size class to the study species. However, importantly, Chapter 5 also demonstrates that as the plastic particles become smaller relative to gape size of the fish, the rate of ingestion increases. Therefore, the degree to which fish are negatively impacted is likely to change in relation to the quantity of various size classes of plastic particles available to them for consumption. This suggests that risk is not static throughout the life cycle of the plastics, or organism, creating a moving target for managers interested in mitigating effects on marine species, and stressing the need for plastic-item specific models that focus on high-risk plastics.

6.3 Implications for the management of plastic pollution

The need to manage inputs, and to reduce the impact of plastics already in the environment, has been internationally recognised. The United Nations has recently signed a resolution for plastic pollution (UNEP, 2017). The resolution is voluntary for countries to adopt and it is legally non-binding, however, it will provide international governments with leverage to design and implement national scale legislation, to manage the issue. In Australia, plastic pollution is recognised as a threat to marine vertebrate life under the Environment Protection and Biodiversity Conservation Act 1999 (EPBC Act), which led to the development of a threat abatement plan for the impacts of marine debris on vertebrate marine life in 2009. This plan, and the EPBC Act 1999, called for coordinated strategies to examine the threats and management options for separate groups of animals. One such plan is the Marine Turtle Recovery Plan (Wallace et al., 2011), which highlights plastic ingestion and entanglement in derelict fishing gear among the threats with considerable knowledge gaps. Individual states within Australia have begun implementing specific legislation to prevent plastic pollution. For example, plastic bag bans exist in many states and will be implemented in Queensland along with a container refund scheme in July 2018 (Queensland Government, 2018). These initiatives are a welcome change to federal and state legislation, however, smaller municipalities have their part to play to reduce plastic pollution in the coastal environment.

The spatial scale of data is an important factor to consider when developing or using outputs from modelling for management action. Nash (2014) suggests ecologists set spatial scales relevant to ecological processes when investigating environmental impacts. However, to take action on an environmental impact, such as plastic pollution in the marine environment, management agencies require information at a spatial scale that is useful for their decision-making, i.e. within the jurisdiction at adequate spatial resolution. Managers also require data that allow them to equally evaluate all locations within their jurisdictions. Field data is commonly collected from beach clean-up events and these events are becoming more routine in many countries (e.g. International Coastal Clean Up). Yet they only provide information at the locations sampled and the locations are generally selected based on the capacity of the group organising the clean-up or ease of access, and they may not represent the best areas for collecting management-relevant information. Modelling can predict inputs and sinks for a study region, however, as my results show, modelling, without robust field data may not accurately represent the situation in complex coastal environments, or at the scale the managers require. My thesis demonstrates: (1) that the modelling environment exists to understand sources and dispersal, and there is a definite need for robust field data at local scales; and (2) field data and modelling studies must be used in tandem to provide multiple lines of evidence on which to base decisions.

My results indicate that local interventions are the most effective method to reduce the input of plastic pollution into the local environment of the Whitsundays region. My models demonstrate that the macroplastics appear to be of local origin, most likely from urban water ways (Chapter 4) and thus managing input at local scale is important. This means that local interventions could have tangible results for reducing exposure to species and habitats in localised areas such as the Whitsundays (Chapter 3) to macroplastic pollution. As discussed in Chapter 3 the consequence of an individual macroplastic item may be higher than the consequence of an individual microplastic. In Chapter 5 I show that low consumption of microplastics result in no significant decline in health to planktivorous reef fish species. This finding is in agreement with Tosetto et al., (2017) who found negligible response to fish personality with plastic consumption via tropic transfer, and literature is emerging detailing the limited impact of microplastics at the current environmental levels (e.g. Kaposi et al., 2014; Davarpanah and Guilhermino, 2015; Ferreira et al., 2016). There is ever-growing literature showing the negative impact of microplastics to the fitness of organisms, however, often the concentrations used are far higher than those currently found in nature and the beads/fragments used are far smaller than size distributions commonly found (Lenz et al., 2016). However, it is logical that if input of plastics into the marine environment continues at the current rate, concentrations found in the habitats will increase and, with degradation, could eventually reach the levels and sizes that will have been shown to have a negative impact to organisms (Cole et al., 2015; Gall and Thompson, 2015; Heindler et al., 2017). To limit the future increase of microplastics in the environment, an effective method is to reduce/remove the inputs and current load of macroplastic in the environment. Doing this will reduce the volume of microplastics arising from macroplastic degradation.

The reduction of input of primary microplastics (those particles that are <5 mm by design) is also desirable. There are methods to reduce the input of primary microplastics. For example, microbead bans, which require production companies to discontinue use of plastic beads in personal care products, have been implemented in many countries (Xanthos and Walker, 2017). Other options lie in technological improvements such as: 1) increasing the capacity of waste-water treatment facilities to remove fibres and beads from waste-water; and 2) designing mechanisms for domestic appliances to remove fibres from waste-water before it leaves the home. The only options that could be implemented at local government scale is to improve waste-treatment facilities, and to increase informed decisions at a consumer level, through education programs. However, with limited funds for management projects I suggest a focus on tackling macroplastic pollution. Action to prevent macroplastics is achievable. Reducing macroplastic pollution would have noticeable impact on local habitats, and would have a knock-on effect of reducing future microplastic pollution.

It is generally accepted that the most effective method of reducing macroplastic input from local sources is with local behaviour change initiatives. Behaviour change science is developing rapidly in the literature and there is strong evidence of valuable contributions and reductions in littering where behaviour change schemes have been undertaken (Benckendorff et al., 2012; Boteler et al., 2015; Spehr & Curnow, 2015; Eagle et al., 2016). Outside of littering, government initiatives, such as the container deposit scheme (Lifset et al., 2013), have proven to reduce plastic waste in the environment. The most effective method of reducing the amount of plastic pollution in the environment is with Integrated Solid Waste Management. This encompasses waste prevention, mitigation, and remediation strategies (see Modak, 2010). Prevention strategies include redesigning products and packaging to minimise waste, or initiating legislation to remove products from the waste stream, e.g. plastic bag bans. Mitigation strategies include recycling and disposal programmes, including container deposit schemes. Lastly, remediation strategies that aim to remove plastic pollution from the environment, including beach cleaning actives or automated systems such as waterwheels (http://baltimorewaterfront.com/healthy-harbor/water-wheel/) and other removal methods (e.g. http://seabinproject.com/, or gross pollutant traps), have been shown to be effective for removing plastic pollution and in some cases other contaminants from the environment. Each of these can be implemented at localised management scales.

The Whitsundays region has many programs already in place to remove plastics from the environment and educate locals and tourists about plastic pollution. The pollution sources identified in this thesis could provide evidence for further specific solutions to be added to increase the impact of these measures. One example, would be to form a strategic partnership with the town of Mackay, to the south of the Whitsundays management jurisdiction, which is a likely source of macroplastics, to incorporate pollution traps at the mouth of the river and storm drains in both management regions. Gross pollutant traps on all storm water outflows in the area could have significant influence on the amount of debris found on Whitsunday region beaches. Lastly, partnering with tourism operators to educate visitors that hire boats and jet skis to conduct independent tours of the area, to make sure they understand the consequences of disposing of waste while at sea.

6.4 Key Limitations

In this section I discuss some of the key limitation to the approach used in this thesis, I discuss the limitations of each chapter in the appropriate section within Chapters 2-5. Through this thesis I have made significant developments in the field of plastic pollution risk and dispersal modelling. Any such development spawns new areas of understanding, and new avenues of research. There are sources of variability at all scales in predicting movement and accumulation of plastic pollution that may influence the model result. In an effort to include parameters that reflect processes that influence plastic dispersal in the natural world, I have included a number of parameters without local-based empirical estimates in the modelling. One example is the degradation parameter, which captures the time taken for macroplastics to degrade, and, in reality, is likely to be variable and dependent on polymer type, UV exposure, temperature, and biofouling. Each of these factors in turn, influences one or more of the others. This can be seen in the case of biofouling, which can reduce the irradiance experienced by the plastic surface, and some polymer types are more sensitive to UV than others (Weinstein et al., 2016). The time scale I used in my modelling approach was deemed applicable based on published literature (Weinstein et al., 2016). However, the degradation process actually produces smaller and smaller microplastics over a presently unknown amount of time. Determining the patterns of degradation of common plastic polymers when at sea and when on beaches would make a useful future research project. In my modelling, one macroplastic degrades into one microplastic and I suggest that future studies consider a more conservative degradation time, with one macroplastic creating many microplastics.

Though this thesis makes good estimations with the incorporation of the wind shadow and reduction of resuspension in areas that are included in the wind shadow. A key limitation of the model is that wave dynamics are not explicitly taken into account. A wave field could be used in the calculation and prediction of resuspension of plastics from the coast (Isobe et al., 2014). To incorporate a wave field into SLIM would have required advanced manipulation of the SLIM source code, which was outside the scope of this thesis. Without a wave parameter it is difficult to accurately parameterise coastal processes, with affect the accumulation of plastics on the shoreline. This parameter is less important if the research project is concerned with suspended plastics.

Another such source of variability is the wind-drift that acts on buoyant objects at or on the surface of the ocean. This parameter is often neglected in modelling plastic distributions (Isobe et al., 2009; Martinez et al., 2009; Kako et al., 2011; Reisser et al., 2013; Isobe et al., 2014; Maes and Blanke, 2015). I used a value of 2% of the wind vector as an average of the values and this is based on literature (Ebbesmeyer et al., 2011; Maximenko et al., 2015), and from flume experiments conducted at JCU's engineering department as part of a set of honours theses (Dunkin, 2015; Gampe, 2015). The wind-drift experienced in nature by plastic particles is dependent on the cross-sectional area of the object exposed to the wind, which is influenced by the object's size, shape, buoyancy, the angle at which the object floats in the water, and if the object is steady or moves (as would occur with a plastic bottle which is partially filled) while drifting (Daniel et al., 2002). All of these factors are not consistent across all plastic objects in the marine environment. Instead, each object would experience a slightly different wind-drift coefficient acting on it. The average wind-drift value used in the modelling is adequate (Dunkin, 2015) and based on available data, however, to accurately represent the variability in plastic pollution it would be necessary to use a range of values in future modelling studies. This could be achieved by creating a rule in the modelling that a proportion of the particles released at the start of each simulation are driven with a different wind-drift value.

The time that the particles are left to mix at the beginning of the simulation, known as the advection time (or integration time) has been found to be a very important factor in the modelling of dispersive objects (Mansui et al., 2015). Therefore, another source of variability in the modelling is the simulation length. All my simulation lengths were chosen to be comparable to one another, but if I had conducted longer simulations the results may have differed. The results of Mansui et al., (2015) show a marked difference in the model output after three months compared to one year. However, the results are not directly applicable to my study area, because their study was conducted at the scale of the whole Mediterranean Sea which is considered to be a semi-closed to closed system. In contrast, the Whitsundays region of the Great Barrier Reef is much smaller and could be considered a semi-open system because water can flush though the study area in either direction. Indeed, in Chapter 4 I found large variation in the daily estimates of plastic accumulation. In future projects, multiple time scales should be considered. This could be achieved by incorporating multiple release times throughout the simulation period and running the scenario for longer durations.

The risk posed by plastics to the marine environment is also dependent on the variability in the consequence of the interaction. The variability in consequence has many origins, for example the species, habitat and plastic type involved in the interaction. For animals, the size of the plastic compared with size of the animal's gape, their foraging behaviour (e.g. surface or benthic foragers) or normal diet items, is important as it dictates the type and intensity of interaction (e.g. entanglement or ingestion). The size and type of the particles may also affect the ability of the animal to recognise plastics as a non-food type or dangerous. Plus, many animals would not learn from the putatively negative experiences of eating and passing plastic particles (Chapter 5). The learning experience is also influenced by the rate at which the animal interacts with the plastic, and this is further dependent on the concentration of the specific plastic type (colour, size, shape) in the habitat used by the animal. The interactions with habitats are less complex and are dependent on the rate of accumulation and the concentration thresholds of negative impact. The interactions with habitats are likely to vary in time (Chapter 3), with the exception of coastline habitats, which were shown in Chapter 3 (Figure 3.12) to have consistent areas of accumulation through time. Based on the results of this thesis the risk that would be most predictable (have least uncertainty) is the risk of macroplastics to coastline habitats, for example mangrove forests.

6.5 Opportunities for future research

This thesis highlights many new directions for research on plastic dispersal. The most prominent is the need to obtain parameter values for the plastics processes included in the model. These values could be obtained through experimental or field-based observation. The variability in these parameter values for each type of plastic commonly found in the environment is also important to accurately model the dispersal of plastic pollution as a whole. This represents a large workload, which could be stratified by first focussing on the types of plastics that most often inflict environmental harm. Additional data on the input from external sources, for example, large cities further south of the study site, as well as shipping lanes, and at a large scale, the input of plastics to the GBR from the Coral Sea, could be used to ground-truth the modelling in a useful way. This could be achieved by incorporating the SLIM GBR model into a larger-scale Pacific Ocean hydrodynamic model that could predict the rate of flux of plastic from the Coral Sea, or south Queensland regions into, and then through, the GBR. Obtaining values for these parameters would improve the ability to model plastic dispersal at any point along the Queensland coast.

An important avenue for future research is ground truthing particle models at this fine scale, ocean is drifter buoys which have been used to truth and/or train particle drift models at the scale of the whole ocean (e.g. van Sebille et al., 2012; 2015). However, this method is rarely used at a small scale, such as the Whitsunday region. Drift card experiments have been conducted (e.g. Walker and Collins, 1985; Steinke and Ward, 2003) providing start and end points for drifting objects. To ground-truth drift models, modern GPS technology could be used to understand the full drift pattern in fine resolution in a small area.

There remain significant knowledge gaps in the consequences of plastic pollution. The consequence at a population level, and for their habitats, must be understood before an environmental risk assessment can be robust and fully informative. We should also attempt to understand what the exposure categories mean for real world consequences and therefore risk. An example of a project would be to investigate ingestion rates and habitat use in each exposure category for a vulnerable species. For example, with marine turtles, a research project could be designed to explore different ingestion rates across age classes or species and understand the types of plastic most commonly ingested. Another option would be to conduct a water quality project sampling for microplastics at the same time as other contaminants in the coastal environment. This would be especially useful to create a link between convergence zones and other areas of the ocean. These convergence zones can often be observed from remote sensing imagery.

6.6 Concluding remarks

The ubiquitous and emerging nature of plastic pollution makes it difficult to manage at a local scale. There is growing understanding of plastic distribution at ocean and sea scale, however, my thesis provides the first example of local jurisdiction scale distribution predictions. With some modification, this technique can be used in other areas of the Great Barrier Reef or other coasts around the world and could also be developed for other types of pollution impacting within small coastal jurisdictions. As well as benefiting smaller-scale needs, my approach can be scaled up to provide meaningful distributions of plastics at a larger management scale, for example, the whole GBR, for use by a management body with a larger jurisdiction. With plastic production increasing exponentially, this pervasive threat will get worse without targeted management action. My thesis demonstrates that understanding the risks and consequences of plastic pollution at the scale of jurisdictional control requires a spatial multi-dimensional approach and requires multiple lines of evidence. The locations of sources need to be understood, especially for macroplastics, and we need more robust data on the physical processes that act on plastics in the marine environment. The locations of the interactions between plastics with habitats and organisms also change through time, with very few consistencies between seasons. These plastics can also affect organisms in different ways as the plastics break up, which also influences how they disperse and the likelihood of interaction. Altogether, this thesis shows that there are a multitude of processes affecting the fate of plastics in the environment. Physical and biological processes all influence the fate of plastics to varying degrees based on the plastics own properties (density, size, shape, etc.). These processes are all acting together, to spread plastics throughout marine and coastal environments. Collectively, my thesis and the new age of modelling are consistently indicating that the scales need to be smaller, sources need to be known and knowledge of how plastics behave in the water is needed.

References

Acampora, H., Schuyler, Q. A., Townsend, K. A. & Hardesty, B. D., 2014. Comparing Plastic Ingestion in Juvenile and Adult Stranded Short-Tailed Shearwaters (*Puffinus Tenuirostris*) in Eastern Australia. Marine Pollution Bulletin 78**,** 63-68.

Acha, E. M., Mianzan, H. W., Iribarne, O., Gagliardini, D. A., Lasta, C. & Daleo, P., 2003. The Role of the Bottom Salinity Front in Accumulating Debris. Marine Pollution Bulletin 46**,** 197-202.

Adams, C. E., Huntingford, F. A., Krpal, J., Jobling, M. & Burnett, S. J., 1995. Exercise, Agonistic Behaviour and Food Acquisition in Arctic Charr, *Salvelinus Alpinus*. Environmental Biology of Fishes 43**,** 213-218.

Adams, C. E., Huntingford, F. A., Turnbull, J. F. & Beattie, C., 1998. Alternative Competitive Strategies and the Cost of Food Acquisition in Juvenile Atlantic Salmon (*Salmo Salar*). Aquaculture 167**,** 17-26.

Anderson, J. C., Park, B. J. & Palace, V. P., 2016. Microplastics in Aquatic Environments: Implications for Canadian Ecosystems. Environmental Pollution 218**,** 269-280.

Andrady, A. L., 2011. Microplastics in the Marine Environment. Marine Pollution Bulletin 62**,** 1596-1605.

Andutta, F. P., Ridd, P. V. & Wolanski, E., 2011. Dynamics of Hypersaline Coastal Waters in the Great Barrier Reef. Estuarine, Coastal and Shelf Science 94**,** 299-305.

Andutta, F. P., Ridd, P. and Wolanski, E., 2013. The age and the flushing time of the Great Barrier Reef waters. Continental Shelf Research 53**,** 11-19.

Andutta, F. P., Kingsford, M. J. & Wolanski, E., 2012. 'Sticky Water'enables the Retention of Larvae in a Reef Mosaic. Estuarine, Coastal and Shelf Science 101**,** 54-63.

Andutta, F. P., Ridd, P. & Wolanski, E., 2013. The Age and the Flushing Time of the Great Barrier Reef Waters. Continental Shelf Research 53**,** 11-19.

Askey, P. J., Richards, S. A., Post, J. R. & Parkinson, E. A., 2006. Linking Angling Catch Rates and Fish Learning under Catch-and-Release Regulations. North American Journal of Fisheries Management 26**,** 1020-1029.

Bakir, A., Rowland, S. J. & Thompson, R. C., 2012. Competitive Sorption of Persistent Organic Pollutants onto Microplastics in the Marine Environment. Marine Pollution Bulletin 64**,** 2782-2789.

Ballent, A., Pando, S., Purser, A., Juliano, M. & Thomsen, L., 2013. Modelled Transport of Benthic Marine Microplastic Pollution in the Nazaré Canyon. Biogeosciences 10**,** 7957.

Ban, N. C., 2009. Minimum Data Requirements for Designing a Set of Marine Protected Areas, Using Commonly Available Abiotic and Biotic Datasets. Biodiversity and Conservation 18**,** 1829-1845.

Barnes, D. K. A., 2002. Biodiversity: Invasions by Marine Life on Plastic Debris. Nature 416**,** 808-809.

Barrows, A. P. W., Neumann, C. A., Berger, M. L. & Shaw, S. D., 2016. Grab Vs. Neuston Tow Net: A Microplastic Sampling Performance Comparison and Possible Advances in the Field. Analytical Methods.

Bates,D., Maechler, M., Bolker, B., Walker S., 2015. Fitting Linear Mixed-Effects Models Using lme4. Journal of Statistical Software, 67, 1-48.<doi:10.18637/jss.v067.i01>.

Baulch, S. & Perry, C., 2014. Evaluating the Impacts of Marine Debris on Cetaceans. Marine Pollution Bulletin 80**,** 210-221.

Beegle-Krause, J. General Noaa Oil Modeling Environment (Gnome): A New Spill Trajectory Model. International Oil Spill Conference, 2001. American Petroleum Institute, 865-871.

Benckendorff, P., Moscardo, G. & Murphy, L., 2012. Environmental Attitudes of Generation Y Students: Foundations for Sustainability Education in Tourism. Journal of Teaching in Travel & Tourism 12, 44-69.

Bengfort, M., Feudel, U., Hilker, F. M. & Malchow, H., 2014. Plankton Blooms and Patchiness Generated by Heterogeneous Physical Environments. Ecological Complexity 20, 185-194.

Besseling, E., Wegner, A., Foekema, E. M., Van Den Heuvel-Greve, M. J. & Koelmans, A. A., 2013. Effects of Microplastic on Fitness and Pcb Bioaccumulation by the Lugworm *Arenicola Marina* (L.). Environmental Science & Technology 47, 593-600.

Biddle, T. M. & Limpus, C. 2011. Marine Wildlife Stranding and Mortality Databade Annual Reports 2005-2010 Marine Turtles. *Conservation Technical and Data Report 2010.*

Bilkovic, D. M., Havens, K., Stanhope, D. & Angstadt, K., 2014. Derelict Fishing Gear in Chesapeake Bay, Virginia: Spatial Patterns and Implications for Marine Fauna. Marine Pollution Bulletin 80, 114-123.

Blasi, M. F. & Mattei, D., 2017. Seasonal Encounter Rate, Life Stages and Main Threats to the Loggerhead Sea Turtle (*Caretta Caretta*) in the Aeolian Archipelago (Southern Thyrrenian Sea). Aquatic Conservation: Marine and Freshwater Ecosystems 27, 617-630.

Boerger, C. M., Lattin, G. L., Moore, S. L. and Moore, C. J., 2010. Plastic ingestion by planktivorous fishes in the North Pacific Central Gyre. Marine Pollution Bulletin 60, 2275-2278.

Boteler, B., Abhold, K., Oosterhuis, F., Fernandez, P., Hadzhiyska, D., Pavlova, D. & Mira Veiga, J., 2015. Best Practice Examples of Existing Economic Policy Instruments and Potential New Economic Policy Instruments to Reduce Marine Litter and Eliminate Barriers to Ges.

Bottrill, M. C., Joseph, L. N., Carwardine, J., Bode, M., Cook, C., Game, E. T., Grantham, H., Kark, S., Linke, S., Mcdonald-Madden, E., Pressey, R. L., Walker, S., Wilson, K. A. & Possingham, H. P., 2008. Is Conservation Triage Just Smart Decision Making? Trends in Ecology & Evolution 23, 649-654.

Boyce, D. G., Lewis, M. R. & Worm, B., 2010. Global Phytoplankton Decline over the Past Century. Nature 466, 591-596.

Bravo Rebolledo, E. L., Van Franeker, J. A., Jansen, O. E. & Brasseur, S. M. J. M., 2013. Plastic Ingestion by Harbour Seals (*Phoca Vitulina*) in the Netherlands. Marine Pollution Bulletin 67, 200-202.

Breivik, Ø., Allen, A. A., Maisondieu, C., & Roth, J. C., 2011. Wind-induced drift of objects at sea: The leeway field method. Applied Ocean Research 33, 100-109.

Brown, C. J., Smith, S. J., Lawton, P. & Anderson, J. T., 2011. Benthic Habitat Mapping: A Review of Progress Towards Improved Understanding of the Spatial Ecology of the Seafloor Using Acoustic Techniques. Estuarine, Coastal and Shelf Science 92, 502-520.

Browne, M. A., Crump, P., Niven, S. J., Teuten, E., Tonkin, A., Galloway, T. & Thompson, R., 2011. Accumulation of Microplastic on Shorelines Woldwide: Sources and Sinks. Environmental Science and Technology 45, 9175-9.

Browne, M. A., Galloway, T. & Thompson, R., 2007. Microplastic—an Emerging Contaminant of Potential Concern? Integrated Environmental Assessment and Management 3, 559-561.

Cable, R. N., Beletsky, D., Beletsky, R., Wigginton, K., Locke, B. W. & Duhaime, M. B., 2017. Distribution and Modeled Transport of Plastic Pollution in the Great Lakes, the World's Largest Freshwater Resource. Frontiers in Environmental Science 5.

Calenge, C. 2011. Home Range Estimation in R: The Adehabitathr Package. Office national de la classe et de la faune sauvage, Saint Benoist, Auffargis, France.

Carpenter, E. J., Anderson, S. J., Harvey, G. R., Miklas, H. P. & Peck, B. B., 1972. Polystyrene Spherules in Coastal Waters. Science 178, 749-750.

Carson, H. S., Colbert, S. L., Kaylor, M. J. & Mcdermid, K. J., 2011. Small Plastic Debris Changes Water Movement and Heat Transfer through Beach Sediments. Marine Pollution Bulletin 62, 1708-1713.

Carson, H. S., Lamson, M. R., Nakashima, D., Toloumu, D., Hafner, J., Maximenko, N. & Mcdermid, K. J., 2013. Tracking the Sources and Sinks of Local Marine Debris in Hawai'I. Marine Environmental Research 84**,** 76-83.

Castañeda, R. A., Avlijas, S., Simard, M. A. & Ricciardi, A., 2014. Microplastic Pollution in St. Lawrence River Sediments. Canadian Journal of Fisheries and Aquatic Sciences 71**,** 1767-1771.

Chae, Y. & An, Y.-J., 2017. Effects of Micro- and Nanoplastics on Aquatic Ecosystems: Current Research Trends and Perspectives. Marine Pollution Bulletin 124**,** 624-632.

Chao, X., Shankar, N. J. and Cheong, H. F., 2001. Two- and three-dimensional oil spill model for coastal waters. Ocean Engineering 28**,** 1557-1573.

Chellappa, S., Huntingford, F. A., Strang, R. H. C. & Thomson, R. Y., 1995. Condition Factor and Hepatosomatic Index as Estimates of Energy Status in Male Three-Spined Stickleback. Journal of Fish Biology 47**,** 775-787.

Chen, C.-L. & Liu, T.-K., 2013. Fill the Gap: Developing Management Strategies to Control Garbage Pollution from Fishing Vessels. Marine Policy 40**,** 34-40.

Chiappone, M., Dienes, H., Swanson, D. W. & Miller, S. L., 2005. Impacts of Lost Fishing Gear on Coral Reef Sessile Invertebrates in the Florida Keys National Marine Sanctuary. Biological Conservation 121**,** 221-230.

Chua, E. M., Shimeta, J., Nugegoda, D., Morrison, P. D. & Clarke, B. O., 2014. Assimilation of Polybrominated Diphenyl Ethers from Microplastics by the Marine Amphipod, *Allorchestes Compressa*. Environmental Science & Technology 48**,** 8127-8134.

Chubarenko, I., Esiukova, E., Bagaev, A., Isachenko, I., Demchenko, N., Zobkov, M., Efimova, I., Bagaeva, M., Khatmullina, L., 2018. Behavior of Microplastics in Coastal Zones. In E. Y. Zeng (Ed.), Microplastic Contamination in Aquatic Environments: An Emerging Matter of Environmental Urgency. pp. 175-223. Cambridge, USA: Elsevier.

Claereboudt, M. R., 2004. Shore Litter Along Sandy Beaches of the Gulf of Oman. Marine Pollution Bulletin 49**,** 770-777.

Claessens, M., Van Cauwenberghe, L., Vandegehuchte, M. B. & Janssen, C. R., 2013. New Techniques for the Detection of Microplastics in Sediments and Field Collected Organisms. Marine Pollution Bulletin 70**,** 227-233.

Colborn, T., Vom Saal, F. S. & Soto, A. M., 1993. Developmental Effects of Endocrine-Disrupting Chemicals in Wildlife and Humans. Environmental Health Perspectives 101**,** 378-384.

Cole, M., Lindeque, P., Fileman, E., Halsband, C. & Galloway, T. S., 2015. The Impact of Polystyrene Microplastics on Feeding, Function and Fecundity in the Marine Copepod *Calanus Helgolandicus*. Environmental Science & Technology 49**,** 1130-1137.

Cole, M., Lindeque, P., Fileman, E., Halsband, C., Goodhead, R., Moger, J. & Galloway, T. S., 2013. Microplastic Ingestion by Zooplankton. Environment Science Technology 47**,** 6646-55.

Colferai, A. S., Silva-Filho, R. P., Martins, A. M. & Bugoni, L., 2017. Distribution Pattern of Anthropogenic Marine Debris Along the Gastrointestinal Tract of Green Turtles (*Chelonia Mydas*) as Implications for Rehabilitation. Marine Pollution Bulletin.

Coll, C., Notter, D., Gottschalk, F., Sun, T., Som, C. & Nowack, B., 2016. Probabilistic Environmental Risk Assessment of Five Nanomaterials (Nano-Tio2, Nano-Ag, Nano-Zno, Cnt, and Fullerenes). Nanotoxicology 10**,** 436-444.

Cooper, D. A. and Corcoran, P. L., 2010. Effects of mechanical and chemical processes on the degradation of plastic beach debris on the island of Kauai, Hawaii. Marine Pollution Bulletin 60**,** 650-654.

Critchell, K., Grech, A., Schlaefer, J., Andutta, F. P., Lambrechts, J., Wolanski, E. & Hamann, M., 2015. Modelling the Fate of Marine Debris Along a Complex Shoreline: Lessons from the Great Barrier Reef. Estuarine, Coastal and Shelf Science 167**,** 414-426.

Cucco, A. & Daniel, P. 2016. Numerical Modeling of Oil Pollution in the Western Mediterranean Sea. Berlin, Heidelberg: Springer.

Daigle, R. M., Metaxas, A. & Deyoung, B., 2014. Fine-Scale Distribution and Spatial Variability of Benthic Invertebrate Larvae in an Open Coastal Embayment in Nova Scotia, Canada. PloS one 9**,** e106178.

Dameron, O. J., Parke, M., Albins, M. A. & Brainard, R., 2007. Marine Debris Accumulation in the Northwestern Hawaiian Islands: An Examination of Rates and Processes. Marine Pollution Bulletin 54**,** 423-433.

Daniel, P., Jan, G., Cabioc'h, F., Landau, Y. & Loiseau, E., 2002. Drift Modeling of Cargo Containers. Spill Science & Technology Bulletin 7**,** 279-288.

Darmon, G., Miaud, C., Claro, F., Doremus, G. & Galgani, F., 2017. Risk Assessment Reveals High Exposure of Sea Turtles to Marine Debris in French Mediterranean and Metropolitan Atlantic Waters. Deep Sea Research Part II: Topical Studies in Oceanography 141**,** 319-328.

Davarpanah, E. & Guilhermino, L., 2015. Single and Combined Effects of Microplastics and Copper on the Population Growth of the Marine Microalgae *Tetraselmis Chuii*. Estuarine, Coastal and Shelf Science 167**,** 269-275.

David, B. O., Closs, G. P., Crow, S. K. & Hansen, E. A., 2007. Is Diel Activity Determined by Social Rank in a Drift-Feeding Stream Fish Dominance Hierarchy? Animal Behaviour 74**,** 259-263.

Dawson, J. L., & Smithers, S. G. (2010). Shoreline and beach volume change between 1967 and 2007 at Raine Island, Great Barrier Reef, Australia. Global and Planetary Change, 72(3), 141-154.

Delandmeter, P., Lewis, S. E., Lambrechts, J., Deleersnijder, E., Legat, V. & Wolanski, E., 2015. The Transport and Fate of Riverine Fine Sediment Exported to a Semi-Open System. Estuarine, Coastal and Shelf Science 167**,** 336-346.

Deleersnijder, E., 2015. A Depth-Integrated Diffusion Problem in a Depth-Varying, Unbounded Domain for Assessing Lagrangian Schemes.

Derraik, J. G. B., 2002. The Pollution of the Marine Environment by Plastic Debris: A Review. Marine Pollution Bulletin 44**,** 842-852.

Desforges, J.-P. W., Galbraith, M., Dangerfield, N. & Ross, P. S., 2014. Widespread Distribution of Microplastics in Subsurface Seawater in the Ne Pacific Ocean. Marine Pollution Bulletin 79**,** 94-99.

Di Bello, A., Valastro, C., Freggi, D., Lai, O. R., Crescenzo, G. & Franchini, D., 2013. Surgical Treatment of Injuries Caused by Fishing Gear in the Intracoelomic Digestive Tract of Sea Turtles. Diseases of Aquatic Organisms 106**,** 93-102.

Do Sul, J. A. I., Costa, M. F., Silva-Cavalcanti, J. S. & Araújo, M. C. B., 2014. Plastic Debris Retention and Exportation by a Mangrove Forest Patch. Marine Pollution Bulletin 78**,** 252-257.

Donohue, M. J., Boland, R. C., Sramek, C. M. & Antonelis, G. A., 2001. Derelict Fishing Gear in the Northwestern Hawaiian Islands: Diving Surveys and Debris Removal in 1999 Confirm Threat to Coral Reef Ecosystems. Marine Pollution Bulletin 42**,** 1301-1312.

Duhec, A. V., Jeanne, R. F., Maximenko, N. & Hafner, J., 2015. Composition and Potential Origin of Marine Debris Stranded in the Western Indian Ocean on Remote Alphonse Island, Seychelles. Marine Pollution Bulletin 96**,** 76-86.

Dunbrack, R. L., Clarke, L. & Bassler, C., 1996. Population Level Differences in Aggressiveness and Their Relationship to Food Density in a Stream Salmonid (*Salvelinus Fontinalis*). Journal of Fish Biology 48**,** 615-622.

Duncan, E. M., Botterell, Z. L. R., Broderick, A. C., Galloway, T. S., Lindeque, P. K., Nuno, A. & Godley, B. J., 2017. A Global Review of Marine Turtle Entanglement in Anthropogenic Debris: A Baseline for Further Action. Endangered Species Research 34**,** 431-448.

Dunkin, M. 2015. A Comparison of Analytical, Experimental, and Numerical Formulations of the Wind-drift Coefficient for Buoyant Marine Litter. Bachelor of Engineering (Mechanical Engineering), James Cook University.
Eagle, L., Hamann, M. & Low, D. R., 2016. The Role of Social Marketing, Marine Turtles and Sustainable Tourism in Reducing Plastic Pollution. Marine Pollution Bulletin 107**,** 324-332.
Ebbesmeyer, C. C., Ingraham, W. J., Royer, T. C. & Grosch, C. E., 2007. Tub Toys Orbit the Pacific Subarctic Gyre. Eos, Transactions American Geophysical Union 88**,** 1-4.
Ebbesmeyer, C. C., Ingraham, W. J., Jones, J. A. & Donohue, M. J., 2011. Marine Debris from the Oregon Dungeness Crab Fishery Recovered in the Northwestern Hawaiian Islands: Identification and Oceanic Drift Paths. Marine Pollution Bulletin 65**,** 69-75.
Eerkes-Medrano, D., Thompson, R. C. & Aldridge, D. C., 2015. Microplastics in Freshwater Systems: A Review of the Emerging Threats, Identification of Knowledge Gaps and Prioritisation of Research Needs. Water Research 75**,** 63-82.
Elliott, J. M., 1973. The Food of Brown and Rainbow Trout (*Salmo Trutta* and *S. Gairdneri*) in Relation to the Abundance of Drifting Invertebrates in a Mountain Stream. Oecologia 12**,** 329-347.
Elliott, J. M., 1976. The Energetics of Feeding, Metabolism and Growth of Brown Trout (*Salmo Trutta L.*) in Relation to Body Weight, Water Temperature and Ration Size. Journal of Animal Ecology 45**,** 923-948.
Emery, A. R., 1973. Comparative Ecology and Functional Osteology of Fourteen Species of Damselfish (Pisces: Pomacentridae) at Alligator Reef, Florida Keys. Bulletin of Marine Science 23**,** 649-770.
Eriksen, M., Lebreton, L. C. M., Carson, H. S., Thiel, M., Moore, C. J., Borerro, J. C., Galgani, F., Ryan, P. G. & Reisser, J., 2014. Plastic Pollution in the World's Oceans: More Than 5 Trillion Plastic Pieces Weighing over 250,000 Tons Afloat at Sea. PloS one 9**,** e111913.
Farrell, P. & Nelson, K., 2013. Trophic Level Transfer of Microplastic: *Mytilus Edulis* (L.) to *Carcinus Maenas* (L.). Environmental Pollution 177**,** 1-3.
Fazey, F. M. C., & Ryan, P. G., 2016. Debris size and buoyancy influence the dispersal distance of stranded litter. Marine Pollution Bulletin 110, 371-377.
Fendall, L. S. & Sewell, M. A., 2009. Contributing to Marine Pollution by Washing Your Face: Microplastics in Facial Cleansers. Marine Pollution Bulletin 58**,** 1225-1228.
Ferreira, P., Fonte, E., Soares, M. E., Carvalho, F. & Guilhermino, L., 2016. Effects of Multi-Stressors on Juveniles of the Marine Fish *Pomatoschistus Microps*: Gold Nanoparticles, Microplastics and Temperature. Aquatic Toxicology 170**,** 89-103.
Fleishman, E., Blockstein, D. E., Hall, J. A., Mascia, M. B., Rudd, M. A., Scott, J. M., Sutherland, W. J., Bartuska, A. M., Brown, A. G., Christen, C. A., Clement, J. P., Dellasala, D., Duke, C. S., Eaton, M., Fiske, S. J., Gosnell, H., Haney, J. C., Hutchins, M., Klein, M. L., Marqusee, J., Noon, B. R., Nordgren, J. R., Orbuch, P. M., Powell, J., Quarles, S. P., Saterson, K. A., Savitt, C. C., Stein, B. A., Webster, M. S. & Vedder, A., 2011. Top 40 Priorities for Science to Inform Us Conservation and Management Policy. BioScience 61**,** 290-300.
Foekema, E. M., De Gruijter, C., Mergia, M. T., Van Franeker, J. A., Murk, A. J. & Koelmans, A. A., 2013. Plastic in North Sea Fish. Environmental Science & Technology 47**,** 8818-8824.
Forrester, G. E., 1991. Social Rank, Individual Size and Group Composition as Determinants of Food Consumption by Humbug Damselfish, *Dascyllus Aruanus*. Animal Behaviour 42**,** 701-711.
Fox, C. H., O'hara, P. D., Bertazzon, S., Morgan, K., Underwood, F. E. & Paquet, P. C., 2016. A Preliminary Spatial Assessment of Risk: Marine Birds and Chronic Oil Pollution on Canada's Pacific Coast. Science of The Total Environment 573**,** 799-809.
Fuentes, M.M.P.B., Blackwood, J., Jones, B., Kim, M., Leis, B., Limpus, C.J., Marsh, H., Mitchell, J., Pouzols, F.M., Pressey, R.L. and Visconti, P., 2015. A decision framework for prioritizing

multiple management actions for threatened marine megafauna. Ecological Applications, 25(1), 200-214.
Gall, S. C. & Thompson, R. C., 2015. The Impact of Debris on Marine Life. Marine Pollution Bulletin 92**,** 170-179.
Game, E. T., Kareiva, P. & Possingham, H. P., 2013. Six Common Mistakes in Conservation Priority Setting. Conservation Biology 27**,** 480-485.
Gampe, D. 2015. Generating Wind-drift Coefficient for Plastics in the Ocean. Bachelor of Engineering (Mechanical Engineering), James Cook University.
García-Berthou, E., 1999. Food of Introduced Mosquitofish: Ontogenetic Diet Shift and Prey Selection. Journal of Fish Biology 55**,** 135-147.
Garvey, J. E., Ostrand, K. G. & Wahl, D. H., 2004. Energetics, Predation, and Ration Affect Size-Dependent Growth and Mortality of Fish During Winter. Ecology 85**,** 2860-2871.
Gilbert, D., 2011. Buried by Bad Decisions. Nature 474**,** 275-277.
Goldberg, E. D., 1997. Plasticizing the Seafloor: An Overview. Environmental Technology 18**,** 195-201.
Goncharov A, Rej R, Negoita S, Schymura M, Santiago-Rivera A & Morse G, 2009. Lower Serum Testosterone Associated with Elevated Polychlorinated Biphenyl Concentrations in Native American Men. Environmental Health Perspectives 117**,** 1454–1460.
González Carman, V., Acha, E. M., Maxwell, S. M., Albareda, D., Campagna, C. & Mianzan, H., 2014. Young Green Turtles, *Chelonia Mydas*, Exposed to Plastic in a Frontal Area of the Sw Atlantic. Marine Pollution Bulletin 78**,** 56-62.
Grech, A. & Marsh, H., 2008. Rapid Asessment of Risks to a Mobile Marine Mammal in an Ecosystem-Scale Marine Protected Area. Conservation Biology 22**,** 711-720.
Grech, A., Coles, R. & Marsh, H., 2011. A Broad-Scale Assessment of the Risk to Coastal Seagrasses from Cumulative Threats. Marine Policy 35**,** 560-567.
Gregory, M. R., 2009. Environmental Implications of Plastic Debris in Marine Settings—Entanglement, Ingestion, Smothering, Hangers-on, Hitch-Hiking and Alien Invasions. Philosophical Transactions of the Royal Society B: Biological Sciences 364**,** 2013-2025.
Griffin, D. A., Oke, P. R. & Jones, E. M. 2016. The Search for MH370 and Ocean Surface Drift. CSIRO Oceans and Atmosphere, Australia.
Guo, W. J. & Wang, Y. X., 2009. A Numerical Oil Spill Model Based on a Hybrid Method. Marine Pollution Bulletin 58**,** 726-734.
Hall, N. M., Berry, K. L. E., Rintoul, L. & Hoogenboom, M. O., 2015. Microplastic Ingestion by Scleractinian Corals. Marine Biology 162**,** 725-732.
Halpern, B. S., Frazier, M., Potapenko, J., Casey, K. S., Koenig, K., Longo, C., Lowndes, J. S., Rockwood, R. C., Selig, E. R., Selkoe, K. A. & Walbridge, S., 2015. Spatial and Temporal Changes in Cumulative Human Impacts on the World's Ocean. Nature Communications 6**,** 7615.
Halpern, B. S., Pyke, C. R., Fox, H. E., Chris Haney, J., Schlaepfer, M. A. & Zaradic, P., 2006. Gaps and Mismatches between Global Conservation Priorities and Spending Brechas E Incongruencias Entre Prioridades Y Gasto De Conservación Global. Conservation Biology 20**,** 56-64.
Halpern, B. S., Walbridge, S., Selkoe, K. A., Kappel, C. V., Micheli, F., D'agrosa, C., Bruno, J. F., Casey, K. S., Ebert, C. & Fox, H. E., 2008. A Global Map of Human Impact on Marine Ecosystems. Science 319**,** 948-952.
Hamann, M., Grech, A., Eric, W. & Lambrechts, J., 2011. Modelling the Fate of Marine Turtle Hatchlings. Ecological Modelling 222**,** 1515-1521.
Hardesty, B. D., Harari, J., Isobe, A., Lebreton, L., Maximenko, N., Potemra, J., Van Sebille, E., Vethaak, A. D. & Wilcox, C., 2017. Using Numerical Model Simulations to Improve the Understanding of Micro-Plastic Distribution and Pathways in the Marine Environment. Frontiers in Marine Science 4.

Hardesty, B. D., Lawson, T., Velde, T., Lansdell, M., & Wilcox, C., 2017. Estimating quantities and sources of marine debris at a continental scale. Frontiers in Ecology and the Environment 15, 18-25.
Hardesty, B. D. and Wilcox, C. 2011. Understanding the types, sources and at-sea distribution of marine debris in Australian waters. CSIRO report.
Harvey, E. T., Kratzer, S. & Philipson, P., 2015. Satellite-Based Water Quality Monitoring for Improved Spatial and Temporal Retrieval of Chlorophyll-a in Coastal Waters. Remote Sensing of Environment 158**,** 417-430.
Heemink, A. W., 1990. Stochastic Modelling of Dispersion in Shallow Water. Stochastic Hydrology and Hydraulics 4**,** 161-174.
Heindler, F. M., Alajmi, F., Huerlimann, R., Zeng, C., Newman, S. J., Vamvounis, G. & Van Herwerden, L., 2017. Toxic Effects of Polyethylene Terephthalate Microparticles and Di(2-Ethylhexyl)Phthalate on the Calanoid Copepod, *Parvocalanus Crassirostris*. Ecotoxicology and Environmental Safety 141**,** 298-305.
Hermsen, E., Pompe, R., Besseling, E. & Koelmans, A. A., 2017. Detection of Low Numbers of Microplastics in North Sea Fish Using Strict Quality Assurance Criteria. Marine Pollution Bulletin 122**,** 253-258.
Hinata, H., Mori, K., Ohno, K., Miyao, Y. & Kataoka, T., 2017. An Estimation of the Average Residence Times and Onshore-Offshore Diffusivities of Beached Microplastics Based on the Population Decay of Tagged Meso- and Macrolitter. Marine Pollution Bulletin 122**,** 17-26.
Hirschenhauser, K. & Oliveira, R. F., 2006. Social Modulation of Androgens in Male Vertebrates: Meta-Analyses of the Challenge Hypothesis. Animal Behaviour 71**,** 265-277.
Hobson, E. S. & Chess, J. R., 1977. Trophic Relationships among Fishes and Plankton in the Lagoon at Enewetak Atoll, Marshall Islands. Fishery Bulletin 76**,** 133-153.
Hoogenboom, M. O., Armstrong, J. D., Groothuis, T. G. G. & Metcalfe, N. B., 2013. The Growth Benefits of Aggressive Behavior Vary with Individual Metabolism and Resource Predictability. Behavioral Ecology 24**,** 253-261.
Hopley, D. 1982. The Geomorphology of the Great Barrier Reef : Quaternary Development of Coral Reefs, Wiley, New York.
Hoss, D. & Settle, L. 1990. Ingestion of Plastics by Teleost Fishes. *In:* SHOMURA, R. S. & CODFREY, H. L., eds. Second International Conference on Marine Debris, 1989 Honolulu, Hawaii. USA.
Hrycik, J.M., Chasse, J., Ruddick, B.R., Taggart, C.T., 2013. Dispersal kernel estimation: a comparison of empirical and modelled particle dispersion in a coastal marine system. Estuarine, Coastand and Shelf Science 133, 11–22.
Hu, Z., Pan, D., He, X. & Bai, Y., 2016. Diurnal Variability of Turbidity Fronts Observed by Geostationary Satellite Ocean Color Remote Sensing. Remote Sensing 8**,** 147.
Hughes, N. F., 1992. Ranking of Feeding Positions by Drift-Feeding Arctic Grayling (*Thymallus Arcticus*) in Dominance Hierarchies. Canadian Journal of Fisheries and Aquatic Sciences 49**,** 1994-1998.
Hughes, T. P., Day, J. C. & Brodie, J., 2015. Securing the Future of the Great Barrier Reef. Nature Climate Change 5**,** 508-511.
Hurst, T. P., Spencer, M. L., Sogard, S. M. & Stoner, A. W., 2005. Compensatory Growth, Energy Storage and Behavior of Juvenile Pacific Halibut *Hippoglossus Stenolepis* Following Thermally Induced Growth Reduction. Marine Ecology Progress Series 293**,** 233-240.
Imre, I., Grant, J. W. A. & Keeley, E. R., 2004. The Effect of Food Abundance on Territory Size and Population Density of Juvenile Steelhead Trout (*Oncorhynchus Mykiss*). Oecologia 138**,** 371-378.

Isobe, A., Kako, S. I., Chang, P.-H. & Matsuno, T., 2009. Two-Way Particle-Tracking Model for Specifying Sources of Drifting Objects: Application to the East China Sea Shelf. Journal of Atmospheric and Oceanic Technology 26**,** 1672-1682.

Isobe, A., Kubo, K., Tamura, Y., Kako, S., Nakashima, E. & Fujii, N., 2014. Selective Transport of Microplastics and Mesoplastics by Drifting in Coastal Waters. Marine Pollution Bulletin 89**,** 324-30.

Ivar Do Sul, J. A. & Costa, M. F., 2014. The Present and Future of Microplastic Pollution in the Marine Environment. Environmental Pollution 185**,** 352-364.

Jackson, N. L., Nordstrom, K. F., Smith, D. R. & Saini, S., 2014. Delivery and Movement of Horseshoe Crab Eggs (*Limulus Polyphemus*) in the Breaking Waves and Swash Uprush of an Estuarine Foreshore. Estuarine, Coastal and Shelf Science 136**,** 191-198.

Jaffal, A., Betoulle, S., Biagianti-Risbourg, S., Terreau, A., Sanchez, W. & Paris-Palacios, S., 2015. Heavy Metal Contamination and Hepatic Toxicological Responses in Brown Trout (*Salmo Trutta*) from the Kerguelen Islands. Polar Research 34**,** 22784.

Jambeck, J. R., Geyer, R., Wilcox, C., Siegler, T. R., Perryman, M., Andrady, A., Narayan, R. & Law, K. L., 2015. Plastic Waste Inputs from Land into the Ocean. Science 347**,** 768-771.

Jang, Y. C., Hong, S., Lee, J., Lee, M. J. & Shim, W. J., 2014. Estimation of Lost Tourism Revenue in Geoje Island from the 2011 Marine Debris Pollution Event in South Korea. Marine Pollution Bulletin 81**,** 49-54.

Johnson, S. W., 1989. Deposition, fate, and characteristics of derelict trawl web on an Alaskan beach. Marine Pollution Bulletin 20**,** 164-168.

Johnson, S. W. & Eiler, J. H., 1999. Fate of Radio-Tagged Trawl Web on an Alaskan Beach. Marine Pollution Bulletin 38**,** 136-141.

Kako, S. I., Isobe, A., Seino, S. & Kojima, A., 2010. Inverse Estimation of Drifting-Object Outflows Using Actual Observation Data. Journal of Oceanography 66**,** 291-297.

Kako, S. I., Isobe, A., Magome, S., Hinata, H., Seino, S. & Kojima, A., 2011. Establishment of Numerical Beach-Litter Hindcast/Forecast Models: An Application to Goto Islands, Japan. Marine Pollution Bulletin 62**,** 293-302.

Kamrowski, R.L., Limpus, C., Moloney, J. and Hamann, M., 2012. Coastal light pollution and marine turtles: assessing the magnitude of the problem. Endangered Species Research, 19(1), 85-98.

Kaposi, K. L., Mos, B., Kelaher, B. P. & Dworjanyn, S. A., 2014. Ingestion of Microplastic Has Limited Impact on a Marine Larva. Environmental Science & Technology 48**,** 1638-1645.

Kashiwada, S., 2006. Distribution of nanoparticles in the see-through medaka (*Oryzias latipes*). Environmental health perspectives, 114, 1697.

Kataoka, T., Hinata, H. & Kato, S., 2015. Backwash Process of Marine Macroplastics from a Beach by Nearshore Currents around a Submerged Breakwater. Marine Pollution Bulletin 101**,** 539-548.

Katsanevakis, S., Verriopoulos, G., Nicolaidou, A. & Thessalou-Legaki, M., 2007. Effect of Marine Litter on the Benthic Megafauna of Coastal Soft Bottoms: A Manipulative Field Experiment. Marine Pollution Bulletin 54**,** 771-778.

Kennedy, D. M. & Woodroffe, C. D., 2002. Fringing Reef Growth and Morphology: A Review. Earth-Science Reviews 57**,** 255-277.

Kinan, I. & Cousins, K., 2000. Abundance of Plastic Debris and Ingestion by Albatross on Kure Atoll, North-Western Hawaiian Islands. Marine Ornithology 28.

Klein, C.J., Beher, J., Chaloupka, M., Hamann, M., Limpus, C. and Possingham, H.P., 2017. Prioritization of marine turtle management projects: a protocol that accounts for threats to different life history stages. Conservation Letters, 10(5), 547-554.

Krelling, A. P., Williams, A. T. & Turra, A., 2017. Differences in Perception and Reaction of Tourist Groups to Beach Marine Debris That Can Influence a Loss of Tourism Revenue in Coastal Areas. Marine Policy 85**,** 87-99.

Kubota, M., Takayama, K., Namimoto, D., 2005. Pleading for the use of biodegradable polymers in favour of marine environments and to avoid an asbestos-like problem for the future. Applied Microbiology and biotechnology 67, 469-476.
Lahr, J. & Kooistra, L., 2010. Environmental Risk Mapping of Pollutants: State of the Art and Communication Aspects. Science of The Total Environment 408**,** 3899-907.
Laist, D. W. 1997. Impacts of Marine Debris: Entanglement of Marine Life in Marine Debris Including a Comprehensive List of Species with Entanglement and Ingestion Records. *In:* COE, J. and ROGERS, D. (eds.) *Marine Debris.* Springer New York.
Lamb, J. B., Willis, B. L., Fiorenza, E. A., Couch, C. S., Howard, R., Rader, D. N., True, J. D., Kelly, L. A., Ahmad, A., Jompa, J. & Harvell, C. D., 2018. Plastic Waste Associated with Disease on Coral Reefs. Science 359**,** 460-462.
Lambrechts, J., Hanert, E., Deleersnijder, E., Bernard, P.-E., Legat, V., Remacle, J.-F., and Eric, W., 2008. A multi-scale model of the hydrodynamics of the whole Great Barrier Reef. Esturina, Coastal and Shelf Science 79**,** 143-151.
Lambrechts, J., Humphrey, C., McKinna, L., Gourge, O., Fabricius, K., Mehta, A., Wolanski, E., 2010. The importance of wave-induced bed fluidisation in the fine sediment budget of Cleveland Bay, Great Barrier Reef. Estuarine, Coastal and Shelf Science 89. 154-162.
Lan, D., Liang, B., Bao, C., Ma, M., Xu, Y. & Yu, C., 2015. Marine Oil Spill Risk Mapping for Accidental Pollution and Its Application in a Coastal City. Marine Pollution Bulletin 96**,** 220-225.
Law, K. L., Morét-Ferguson, S. E., Goodwin, D. S., Zettler, E. R., Deforce, E., Kukulka, T. & Proskurowski, G., 2014. Distribution of Surface Plastic Debris in the Eastern Pacific Ocean from an 11-Year Data Set. Environmental Science & Technology 48**,** 4732-4738.
Le Hénaff, M., Kourafalou, V. H., Paris, C. B., Helgers, J., Aman, Z. M., Hogan, P. J. and Srinivasan, A., 2012. Surface evolution of the Deepwater horizon oil spill patch: combined effects of circulation and wind-induced drift. Environmental Science and Technology 46**,** 7267-7273.
Lebreton, L., Greer, S. & Borrero, J., 2012. Numerical Modelling of Floating Debris in the World's Oceans. Marine Pollution Bulletin 64**,** 653-661.
Lebreton, L. & Borrero, J., 2013. Modeling the Transport and Accumulation Floating Debris Generated by the 11 March 2011 Tohoku Tsunami. Marine Pollution Bulletin 66**,** 53-58.
Lee, D.-I., Cho, H.-S., Jeong, S.-B., 2006. Distribution characteristics of marine litter on the sea bed of the East China Sea and the South Sea of Korea. Estuarine, Coastal and Shelf Science 70**,** 187-194.
Lenz, R., Enders, K. & Nielsen, T. G., 2016. Microplastic Exposure Studies Should Be Environmentally Realistic. Proceedings of the National Academy of Sciences 113**,** E4121-E4122.
Li, Y., Brimicombe, A. J. & Ralphs, M. P., 2000. Spatial Data Quality and Sensitivity Analysis in Gis and Environmental Modelling: The Case of Coastal Oil Spills. Computers, Environment and Urban Systems 24**,** 95-108.
Lifset, R., Atasu, A. & Tojo, N., 2013. Extended Producer Responsibility. Journal of Industrial Ecology 17**,** 162-166.
Lima, A. R. A., Barletta, M. & Costa, M. F., 2015. Seasonal Distribution and Interactions between Plankton And microplastics in a Tropical Estuary. Estuarine, Coastal and Shelf Science 165**,** 213-225.
Low, R. M., 1971. Interspecific Territoriality in a Pomacentrid Reef Fish, *Pomacentrus Flavicauda* Whitley. Ecology 52**,** 648-654.
Lusher, A. L., Mchugh, M. & Thompson, R. C., 2013. Occurrence of Microplastics in the Gastrointestinal Tract of Pelagic and Demersal Fish from the English Channel. Marine Pollution Bulletin 67**,** 94-99.

Ma, Z., Guo, H., Zhang, D., Hu, C. & Jiang, S., 2015. Food Ingestion, Consumption and Selectivity of Pompano, *Trachinotus Ovatus* (Linnaeus 1758) under Different Rotifer Densities. Aquaculture Research 46**,** 2593-2603.

Maes, C. & Blanke, B., 2015. Tracking the Origins of Plastic Debris across the Coral Sea: A Case Study from the Ouvéa Island, New Caledonia. Marine Pollution Bulletin 97**,** 160-168.

Mansui, J., Molcard, A. & Ourmières, Y., 2015. Modelling the Transport and Accumulation of Floating Marine Debris in the Mediterranean Basin. Marine Pollution Bulletin 91**,** 249-257.

Mao, Y. & Ridd, P. V., 2015. Sea Surface Temperature as a Tracer to Estimate Cross-Shelf Turbulent Diffusivity and Flushing Time in the Great Barrier Reef Lagoon. Journal of Geophysical Research: Oceans 120**,** 4487-4502.

Margules, C. R. & Pressey, R. L., 2000. Systematic Conservation Planning. Nature 405**,** 243-253.

Martinez, E., Maamaatuaiahutapu, K. & Taillandier, V., 2009. Floating Marine Debris Surface Drift: Convergence and Accumulation toward the South Pacific Subtropical Gyre. Marine Pollution Bulletin 58**,** 1347-1355.

Maximenko, N., Hafner, J. and Niiler, P., 2012. Pathways of marine debris derived from trajectories of lagrangian drifters. Marine Pollution Bulletin 65**,** 51-62.

Maximenko, N., Macfadyen, A. & Kamachi, M., 2015. Modeling the Drift of Marine Debris Generated by the 2011 Tsunami in Japan. PICES Press 23.

McElwee, K., Donohhue, M., Courtney, C., Morishige, C., Rivera-Vicente, A., 2012. A strategy for detecting derelict fishing gear at sea. Marine Pollution Bulletin 65, 7-15.

McKinney, J. A., Hoffmayer, E. R., Wu, W., Fulford, R. & Hendon, J., 2012. Feeding Habitat of the Whale Shark *Rhincodon Typus* in the Northern Gulf of Mexico Determined Using Species Distribution Modelling. Marine Ecology Progress Series 458**,** 199-211.

McPherson, M., Schill, S., Raber, G., John, K., N., Z., Thurlow, K. & Sutton, A. H., 2008. Gis-Based Modeling of Environmental Risk Surfaces (Ers) for Conservation Planning in Jamaica. Journal of Conservation Planning 4**,** 60-89.

Metcalfe, N. B. & Monaghan, P., 2001. Compensation for a Bad Start: Grow Now, Pay Later? Trends in Ecology & Evolution 16**,** 254-260.

Modak, P. 2010. Municipal Solid Waste Management: Turning Waste into Resources. *Shanghai Manual – a Guide for Sustainable Urban Development in the 21st Century.*: United Nations Department of Economic and Social Affairs (UNDESA, 2010).

Moore, C., Lattin, G. & Zellers, A., 2011. Quantity and Type of Plastic Debris Flowing from Two Urban Rivers to Coastal Waters and Beaches of Southern California. Revista da Gestão Costeira Integrada 11**,** 65-73.

Morét-Ferguson, S., Law, K. L., Proskurowski, G., Murphy, E. K., Peacock, E. E. & Reddy, C. M., 2010. The Size, Mass, and Composition of Plastic Debris in the Western North Atlantic Ocean. Marine Pollution Bulletin 60**,** 1873-1878.

Myksvoll, M. S., Sandvik, A. D., Skarðhamar, J. & Sundby, S., 2012. Importance of High Resolution Wind Forcing on Eddy Activity and Particle Dispersion in a Norwegian Fjord. Estuarine, Coastal and Shelf Science 113**,** 293-304.

Naik, M., 2017. 5 Ways Plastics Revolutionized the Healthcare Industry. Medical Product Outsourcing Magazine 10/09/2017.

Nash, H. L., 2014. Research Article: Defining Appropriate Spatial and Temporal Scales for Ecological Impact Analysis. Environmental Practice 16**,** 281-286.

Nelms, S. E., Duncan, E. M., Broderick, A. C., Galloway, T. S., Godfrey, M. H., Hamann, M., Lindeque, P. K. & Godley, B. J., 2016. Plastic and Marine Turtles: A Review and Call for Research. ICES Journal of Marine Science 73**,** 165-181.

Neves, D., Sobral, P., Ferreira, J. L. & Pereira, T., 2015. Ingestion of Microplastics by Commercial Fish Off the Portuguese Coast. Marine Pollution Bulletin 101**,** 119-126.

Nixon, Z. & Barnea, N. 2010. Development of the Gulf of Mexico Marine Debris Model. NOAA Technical Memorandum.

Norton, T., Beer, T. & Dovers, S., 1996. Risk and Uncertainty in Environmental Management. Centre for Resource and Environmental Studies, Australian National University, Canberra.

Okubo, A., 1971. Diffusion diagrams. Deep-Sea Research 18, 789–802.

O'Brien, W. J., Slade, N. A. & Vinyard, G. L., 1976. Apparent Size as the Determinant of Prey Selection by Bluegill Sunfish (*Lepomis Macrochirus*). Ecology 57**,** 1304-1310.

Parga, M. L., 2012. Hooks and Sea Turtles: A Veterinarian's Perspective. Bulletin of Marine Science 88**,** 731-741.

Peng G., Leslie L. M., Shao Y. 2013. Environmental modelling and prediction. Springer Science & Business Media, Berlin.

Pepper, A. & Puotinen, M. L., 2009. Gremo: A GIS-Based Generic Model for Estimating Relative Wave Exposure.

Pereira, J. J., Mercaldo-Allen, R., Kuropat, C., Luedke, D. & Sennefelder, G., 1993. Effect of Cadmium Accumulation on Serum Vitellogenin Levels and Hepatosomatic and Gonadosomatic Indices of Winter Flounder (*Pleuronectes Americanus*). Archives of Environmental Contamination and Toxicology 24**,** 427-431.

Phillips, R. R. & Swears, S. B., 1979. Social Hierarchy, Shelter Use, and Avoidance of Predatory Toadfish (*Opsanus Tau*) by the Striped Blenny (*Chasmodes Bosquianus*). Animal Behaviour 27**,** 1113-1121.

Pichel, W. G., Veenstra, T. S., Churnside, J. H., Arabini, E., Friedman, K. S., Foley, D. G., Brainard, R. E., Kiefer, D., Ogle, S., Clemente-Colón, P. & Li, X., 2012. Ghostnet Marine Debris Survey in the Gulf of Alaska – Satellite Guidance and Aircraft Observations. Marine Pollution Bulletin 65**,** 28-41.

Pierce, K. E., Harris, R. J., Larned, L. S. & Pokras, M. A., 2004. Obstruction and Starvation Associated with Plastic Ingestion in a Northern Gannet *Morus Bassanus* and a Greater Shearwater *Puffinus Gravis*. Marine Ornithology 32**,** 187-189.

Pinheiro, J., Bates D., DebRoy S., Sarkar D. and R Core Team. 2018. nlme: Linear and Nonlinear Mixed Effects Models. R package version 3.1-137, https://CRAN.R-project.org/package=nlme.

Plastics Europe 2016. Plastics - The Facts 2016 an Anlysis of European Plastics Production , Demand and Waste Data. Plastics Europe.

Possatto, F. E., Barletta, M., Costa, M. F., Ivar Do Sul, J. A. & Dantas, D. V., 2011. Plastic Debris Ingestion by Marine Catfish: An Unexpected Fisheries Impact. Marine Pollution Bulletin 62**,** 1098-1102.

Potemra, J. T., 2012. Numerical Modeling with Application to Tracking Marine Debris. Marine Pollution Bulletin 65**,** 42-50.

Queensland Government. 2018. *About the Container Refund Scheme* [Online]. Available: https://www.qld.gov.au/environment/pollution/management/waste/container-refund-about [Accessed 15/01/2018].

R Core Team 2016. R: A Language and Environment for Statistical Computing. Vienna, Austria: R Foundation for Statistical Computing.

Reed, M., Aamo, O. M. & Daling, P. S., 1995. Quantitative Analysis of Alternate Oil Spill Response Strategies Using Oscar. Spill Science & Technology Bulletin 2**,** 67-74.

Reisser, J., Shaw, J., Wilcox, C., Hardesty, B. D., Proietti, M., Thums, M. & Pattiaratchi, C., 2013. Marine Plastic Pollution in Waters around Australia: Characteristics, Concentrations, and Pathways. PloS one 8**,** e80466.

Reisser, J., Slat, B., Noble, K., Du Plessis, K., Epp, M., Proietti, M., De Sonneville, J., Becker, T. & Pattiaratchi, C., 2015. The Vertical Distribution of Buoyant Plastics at Sea: An Observational Study in the North Atlantic Gyre. Biogeosciences 12**,** 1249.

Rochman, C. M., Hoh, E., Hentschel, B. T. & Kaye, S., 2013a. Long-Term Field Measurement of Sorption of Organic Contaminants to Five Types of Plastic Pellets: Implications for Plastic Marine Debris. Environmental Science & Technology 47**,** 1646-1654.
Rochman, C. M., Hoh, E., Kurobe, T. & Teh, S. J., 2013b. Ingested Plastic Transfers Hazardous Chemicals to Fish and Induces Hepatic Stress. Scientific reports 3**,** 3263.
Rochman, C. M., Kurobe, T., Flores, I., The, S. J., 2014. Early Warning Signs of Endocrine Disruption in Adult Fish From The Ingestion of Polythylene with and without Sorbed Chemical Pollutants from the Marine Enviornment. Science of the Total Environment 493, 656-661.
Romeo, T., Pietro, B., Pedà, C., Consoli, P., Andaloro, F. & Fossi, M. C., 2015. First Evidence of Presence of Plastic Debris in Stomach of Large Pelagic Fish in the Mediterranean Sea. Marine Pollution Bulletin 95**,** 358-361.
Rummel, C. D., Löder, M. G. J., Fricke, N. F., Lang, T., Griebeler, E.-M., Janke, M. & Gerdts, G., 2016. Plastic Ingestion by Pelagic and Demersal Fish from the North Sea and Baltic Sea. Marine Pollution Bulletin 102**,** 134-141.
Ryan, J. P., Harvey, J. B. J., Zhang, Y. & Woodson, C. B., 2014. Distributions of Invertebrate Larvae and Phytoplankton in a Coastal Upwelling System Retention Zone and Peripheral Front. Journal of Experimental Marine Biology and Ecology 459**,** 51-60.
Ryan, P. G., 1987. The Incidence and Characteristics of Plastic Particles Ingested by Seabirds. Marine Environmental Research 23**,** 175-206.
Ryan, P. G., 1988. Effects of Ingested Plastic on Seabird Feeding: Evidence from Chickens. Marine Pollution Bulletin 19**,** 125-128.
Ryer, C. H., Stoner, A. W., Iseri, P. J. & Spencer, M. L., 2009. Effects of Simulated Underwater Vehicle Lighting on Fish Behavior. Marine Ecology Progress Series 391**,** 97-106.
Savoca, M. S., Tyson, C. W., Mcgill, M. & Slager, C. J., 2017. Odours from Marine Plastic Debris Induce Food Search Behaviours in a Forage Fish. Proceedings of the Royal Society B: Biological Sciences 284.
Schmidt, C., Krauth, T. & Wagner, S., 2017. Export of Plastic Debris by Rivers into the Sea. Environmental Science & Technology 51**,** 12246-12253.
Schuyler, Q., Hardesty, B. D., Wilcox, C. & Townsend, K., 2012. To Eat or Not to Eat? Debris Selectivity by Marine Turtles. PloS one 7**,** e40884.
Schuyler, Q., Hardesty, B. D., Wilcox, C. & Townsend, K., 2014. Global Analysis of Anthropogenic Debris Ingestion by Sea Turtles. Conservation Biology 28**,** 129-139.
Schuyler, Q. A., Wilcox, C., Townsend, K. A., Wedemeyer-Strombel, K. R., Balazs, G., Van Sebille, E. & Hardesty, B. D., 2016. Risk Analysis Reveals Global Hotspots for Marine Debris Ingestion by Sea Turtles. Global Change Biology 22**,** 567-576.
Schwaiger, J., Wanke, R., Adam, S., Pawert, M., Honnen, W. & Triebskorn, R., 1997. The Use of Histopathological Indicators to Evaluate Contaminant-Related Stress in Fish. Journal of Aquatic Ecosystem Stress and Recovery 6**,** 75-86.
Setälä, O., Fleming-Lehtinen, V. & Lehtiniemi, M., 2014. Ingestion and Transfer of Microplastics in the Planktonic Food Web. Environmental Pollution 185**,** 77-83.
Sherman, P. & Van Sebille, E., 2016. Modeling Marine Surface Microplastic Transport to Assess Optimal Removal Locations. Environmental Research Letters 11**,** 014006.
Shimada, T., Jones, R., Limpus, C. & Hamann, M., 2012. Improving Data Retention and Home Range Estimates by Data-Driven Screening. Marine Ecology Progress Series 457**,** 171-180.
Shimada, T., Jones, R., Limpus, C., Groom, R. & Hamann, M., 2016. Long-Term and Seasonal Patterns of Sea Turtle Home Ranges in Warm Coastal Foraging Habitats: Implications for Conservation. Marine Ecology Progress Series 562**,** 163-179.
Short, A. D. & Woodroffe, C. D., 2009. The Coast of Australia, Cambridge University Press, New York.

Smith, S. D. A., 2012. Marine debris: A proximate threat to marine sustainability in Bootless Bay, Papua New Guinea. Marine Pollution Bulletin 64, 1880-1883.
Smith, S. D. A., Markic, A., 2013. Estimates of Marine Debris Accumulation on Beaches Are Strongly Affected by the Temporal Scale of Sampling. PLOS ONE 8 e83694.
Smith, S. D. A., Gillies, C. L., Shortland-Jones, H., 2014. Patterns of marine debris distribution on the beaches of Rottnest Island, Western Australia. Marine Pollution Bulletin. 88, 188-193.
Smith, S. D. A., Banister, K., Fraser, N., Edgar, R. J., 2018. Tracing the source of marine debris on the beaches of northern New South Wales, Australia: The Bottles on Beaches program. Marine Pollution Bulletin 126, 304-307.
Spagnol, S., Wolanski, E., Deleersnijder, E., Brinkman, R., Mcallister, F., Cushman-Roisin, B. & Hanert, E., 2002. An Error Frequently Made in the Evaluation of Advective Transport in Two-Dimensional Lagrangian Models of Advection-Diffusion in Coral Reef Waters. Marine Ecology Progress Series 235**,** 299-302.
Sparks, R. E., Waller, W. T. & Cairns Jr, J., 1972. Effect of Shelters on the Resistance of Dominant and Submissive Blue Gills (*Lepomis Macrochirus*) to a Lethal Concentration of Zinc. Journal of the Fisheries Research Board of Canada 29**,** 1356-1358.
Spaulding, M. L., 1988. A state-of-the-art review of oil spill trajectory and fate modeling. Oil and Chemical Pollution 4**,** 39-55.
Spaulding, M. L., 2017. State of the Art Review and Future Directions in Oil Spill Modeling. Marine Pollution Bulletin 115**,** 7-19.
Spear, L. B., Ainley, D. G. & Ribic, C. A., 1995. Incidence of Plastic in Seabirds from the Tropical Pacific, 1984–1991: Relation with Distribution of Species, Sex, Age, Season, Year and Body Weight. Marine Environmental Research 40**,** 123-146.
Spehr, K., & Curnow, R., 2015. Litterology: Understanding littering and the secrets to clean public places. Environment Books, Frankston Victoria
Srivastava, P. K., Gupta, M. & Mukherjee, S., 2012. Mapping Spatial Distribution of Pollutants in Groundwater of a Tropical Area of India Using Remote Sensing and GIS. Applied Geomatics 4**,** 21-32.
Staddon, J. E. 2016. Adaptive Behavior and Learning, Cambridge University Press.
Steinke, T. & Ward, C., 2003. Use of Plastic Drift Cards as Indicators of Possible Dispersal of Propagules of the Mangrove *Avicennia Marina* by Ocean Currents. African Journal of Marine Science 25**,** 169-176.
Steinmetz, Z., Wollmann, C., Schaefer, M., Buchmann, C., David, J., Tröger, J., Muñoz, K., Frör, O. & Schaumann, G. E., 2016. Plastic Mulching in Agriculture. Trading Short-Term Agronomic Benefits for Long-Term Soil Degradation? Science of The Total Environment 550**,** 690-705.
Storrier, K. L., Mcglashan, D. J., Bonellie, S. & Velander, K., 2007. Beach Litter Deposition at a Selection of Beaches in the Firth of Forth, Scotland. Journal of Coastal Research**,** 813-822.
StrandNet, Queensland Department of Environment and Science Marine Wildlife Stranding Database, https://www.ehp.qld.gov.au/wildlife/caring-for-wildlife/strandnet-reports.html
Thiel, M., Hinojosa, I. A., Miranda, L., Pantoja, J. F., Rivadeneira, M. M. & Vásquez, N., 2013. Anthropogenic marine debris in the coastal environment: a multi-year comparison between coastal waters and local shores. Marine Pollution Bulletin 71**,** 307-316.
Thompson, R. C., Moore, C. J., Von Saal, F. S. & Swan, S. H., 2009. Plastics, the Environment and Human Health: Current Consensus and Future Trends. Philosophical Transactions of The Royal Society Biological Sciences 1**,** 1-14.

Titmus, A. J. & Hyrenbach, D. K., 2011. Habitat Associations of Floating Debris and Marine Birds in the North East Pacific Ocean at Coarse and Meso Spatial Scales. Marine Pollution Bulletin 62**,** 2496-2506.

Tkalich, P., Huda, K. and Hoong Gin, K. Y., 2003. A multiphase oil spill model. Journal of Hydraulic Research 41**,** 115-125.

Toobaie, A. & Grant, J. W. A., 2013. Effect of Food Abundance on Aggressiveness and Territory Size of Juvenile Rainbow Trout, *Oncorhynchus Mykiss*. Animal Behaviour 85**,** 241-246.

Topçu, E. N., Tonay, A. M., Dede, A., Öztürk, A. A. & Öztürk, B., 2013. Origin and Abundance of Marine Litter Along Sandy Beaches of the Turkish Western Black Sea Coast. Marine Environmental Research 85**,** 21-28.

Tosetto, L., Williamson, J. E. & Brown, C., 2017. Trophic Transfer of Microplastics Does Not Affect Fish Personality. Animal Behaviour 123**,** 159-167.

Tudor, D. & Williams, A., 2004. Development of a 'Matrix Sorting Technique' to Detirmine Litter Sources at a Bristol Channel Beach. Journal of Coastal Conservation 10, 119-127.

Underwood, A., Chapman, M. & Browne, M. A., 2017. Some Problems and Practicalities in Design and Interpretation of Samples of Microplastic Waste. Analytical Methods 9**,** 1332-1345.

UNEP 2002. Stockholm Convention on Persistent Organic Pollutants (POPS) *In:* PROGRAMME, U. N. E. (ed.). Geneva.

Uneputty, P. & Evans, S. M., 1997. The Impact of Plastic Debris on the Biota of Tidal Flats in Ambon Bay (Eastern Indonesia). Marine Environmental Research 44**,** 233-242.

Valdor, P. F., Puente, A., Gómez, A. G., Ondiviela, B. & Juanes, J. A., 2017. Are Environmental Risk Estimations Linked to the Actual Environmental Impact? Application to an Oil Handling Facility (Ne Spain). Marine Pollution Bulletin 114**,** 941-951.

Van Cauwenberghe, L., Claessens, M., Vandegehuchte, M. B. & Janssen, C. R., 2015. Microplastics Are Taken up by Mussels (*Mytilus Edulis*) and Lugworms (*Arenicola Marina*) Living in Natural Habitats. Environmental Pollution 199**,** 10-17.

van Den Berg, H., 2009. Global Status of Ddt and Its Alternatives for Use in Vector Control to Prevent Disease. Environmental Health Perspectives 117**,** 1656-1663.

Van Sebille, E., England, M. & Froyland, G., 2012. Origin, Dynamics and Evolution of Ocean Garbage Patches from Observed Surface Drifters. Environmental Research Letters 7.

Van Sebille, E., Wilcox, C., Lebreton, L., Maximenko, N., Hardesty, B. D., Van Franeker, J. A., Eriksen, M., Siegel, D., Galgani, F. & Law, K. L., 2015. A Global Inventory of Small Floating Plastic Debris. Environmental Research Letters 10**,** 124006.

Vegter, A., Barletta, M., Beck, C., Borrero, J., Burton, H., Campbell, M., Eriksen, M., Eriksson, C., Estrades, A., Gilardi, K., Hardesty, B. D., Sul, J. a. I. D., Lavers, J. L., Lazar, B., Lebreton, L., Nichols, W. J., Ribic, C., Ryan, P., Schuyler, Q. A., Smith, S. D. A., Takada, H., Townsend, K., Wabnitz, C. C. C., Wilcox, C., Young, L. & Hamann, M., 2014. Global Research Priorities to Mitigate Plastic Pollution Impacts on Marine Wildlife. Endangered Species Research 25**,** 225-247.

Vendel, A. L., Bessa, F., Alves, V. E. N., Amorim, A. L. A., Patrício, J. & Palma, A. R. T., 2017 Widespread Microplastic Ingestion by Fish Assemblages in Tropical Estuaries Subjected to Anthropogenic Pressures. Marine Pollution Bulletin. 117, 448-455.

Verlis, K. M., Campbell, M. L. & Wilson, S. P., 2013. Ingestion of Marine Debris Plastic by the Wedge-Tailed Shearwater *Ardenna Pacifica* in the Great Barrier Reef, Australia. Marine Pollution Bulletin 72**,** 244-249.

Walker, T. & Collins, J. Surface Circulation in the Central Region of the Great Barrier Reef Lagoon. the fifth international Coral Reef Congress, 1985 Tahiti. 23-28.

Wallace, B. P., Dimatteo, A. D., Hurley, B. J., Finkbeiner, E. M., Bolten, A. B., Chaloupka, M. Y., Hutchinson, B. J., Abreu-Grobois, F. A., Amorocho, D., Bjorndal, K. A., Bourjea, J., Bowen, B. W., Dueñas, R. B., Casale, P., Choudhury, B. C., Costa, A., Dutton, P. H., Fallabrino, A., Girard, A., Girondot, M., Godfrey, M. H., Hamann, M., López-Mendilaharsu, M., Marcovaldi, M. A., Mortimer, J. A., Musick, J. A., Nel, R., Pilcher, N. J., Seminoff, J. A., Troëng, S., Witherington, B. & Mast, R. B., 2010. Regional Management Units for Marine Turtles: A Novel Framework for Prioritizing Conservation and Research across Multiple Scales. PloS one 5**,** e15465.

Wallace, B. P., Dimatteo, A. D., Bolten, A. B., Chaloupka, M. Y., Hutchinson, B. J., Abreu-Grobois, F. A., Mortimer, J. A., Seminoff, J. A., Amorocho, D., Bjorndal, K. A., Bourjea, J., Bowen, B. W., Briseño Dueñas, R., Casale, P., Choudhury, B. C., Costa, A., Dutton, P. H., Fallabrino, A., Finkbeiner, E. M., Girard, A., Girondot, M., Hamann, M., Hurley, B. J., López-Mendilaharsu, M., Marcovaldi, M. A., Musick, J. A., Nel, R., Pilcher, N. J., Troëng, S., Witherington, B. & Mast, R. B., 2011. Global Conservation Priorities for Marine Turtles. PloS one 6**,** e24510.

Watts, A. J. R., Lewis, C., Goodhead, R. M., Beckett, S. J., Moger, J., Tyler, C. R. & Galloway, T. S., 2014. Uptake and Retention of Microplastics by the Shore Crab *Carcinus Maenas*. Environmental Science & Technology 48**,** 8823-8830.

Weinstein, J. E., Crocker, B. K. & Gray, A. D., 2016. From Macroplastic to Microplastic: Degradation of High-Density Polyethylene, Polypropylene, and Polystyrene in a Salt Marsh Habitat. Environmental Toxicology and Chemistry 35**,** 1632-1640.

Welden, N. A. & Cowie, P. R., 2017. Degradation of Common Polymer Ropes in a Sublittoral Marine Environment. Marine Pollution Bulletin 118**,** 248-253.

Whittock, P. A., Pendoley, K. L. & Hamann, M., 2016. Using Habitat Suitability Models in an Industrial Setting: The Case for Internesting Flatback Turtles. Ecosphere 7**,** e01551-n/a.

Wilcox, C., Hardesty, B. D., Sharples, R., Griffin, D. A., Lawson, T. J. & Gunn, R., 2013. Ghostnet Impacts on Globally Threatened Turtles, a Spatial Risk Analysis for Northern Australia. Conservation Letters 1**,** 1-8.

Wilcox, C., Van Sebille, E. & Hardesty, B. D., 2015. Threat of Plastic Pollution to Seabirds Is Global, Pervasive, and Increasing. Proceedings of the National Academy of Sciences 112**,** 11899-11904.

Wildermann, N. 2017. Flatbacks at Sea: Understanding Ecology in Foraging Populations. Doctor of Philosophy, James Cook University.

Wildermann, N., Critchell, K., Fuentes, M. M. P. B., Limpus, C. J., Wolanski, E. & Hamann, M., 2017. Does Behaviour Affect the Dispersal of Flatback Post-Hatchlings in the Great Barrier Reef? Royal Society Open Science 4.

Willis, K., Maureaud, C., Wilcox, C. & Hardesty, B. D., 2017. How Successful Are Waste Abatement Campaigns and Government Policies at Reducing Plastic Waste into the Marine Environment? Marine Policy.

Wilson, K. A., Mcbride, M. F., Bode, M. & Possingham, H. P., 2006. Prioritizing Global Conservation Efforts. Nature 440**,** 337-340.

Wolanski, E. and Delesalle, B., 1995. Wind-driven upwelling in Opunohu Bay, Moorea, French Polynesia. Estuarine, Coastal and Shelf Science 40**,** 57-66.

Wolanski, E. & Hamner, W. M., 1988. Topographically Controlled Fronts in the Ocean and Their Biological Influence. Science 241**,** 177-181.

Wolanski, E., Jones, M. & Williams, W., 1981. Physical Properties of the Great Barrier Reef Lagoon Waters near Townsville. Ii. Seasonal Variations. Marine and Freshwater Research 32**,** 321-334.

Worm, B., Lotze, H. K., Jubinville, I., Wilcox, C. & Jambeck, J., 2017. Plastic as a Persistent Marine Pollutant. Annual Review of Environment and Resources 42**,** 1-26.

Wright, S. L., Rowe, D., Thompson, R. C. & Galloway, T. S., 2013a. Microplastic Ingestion Decreases Energy Reserves in Marine Worms. Current Biology 23**,** R1031-R1033.

Wright, S. L., Thompson, R. C. & Galloway, T. S., 2013b. The Physical Impacts of Microplastics on Marine Organisms: A Review. Environmental Pollution 178**,** 483-492.

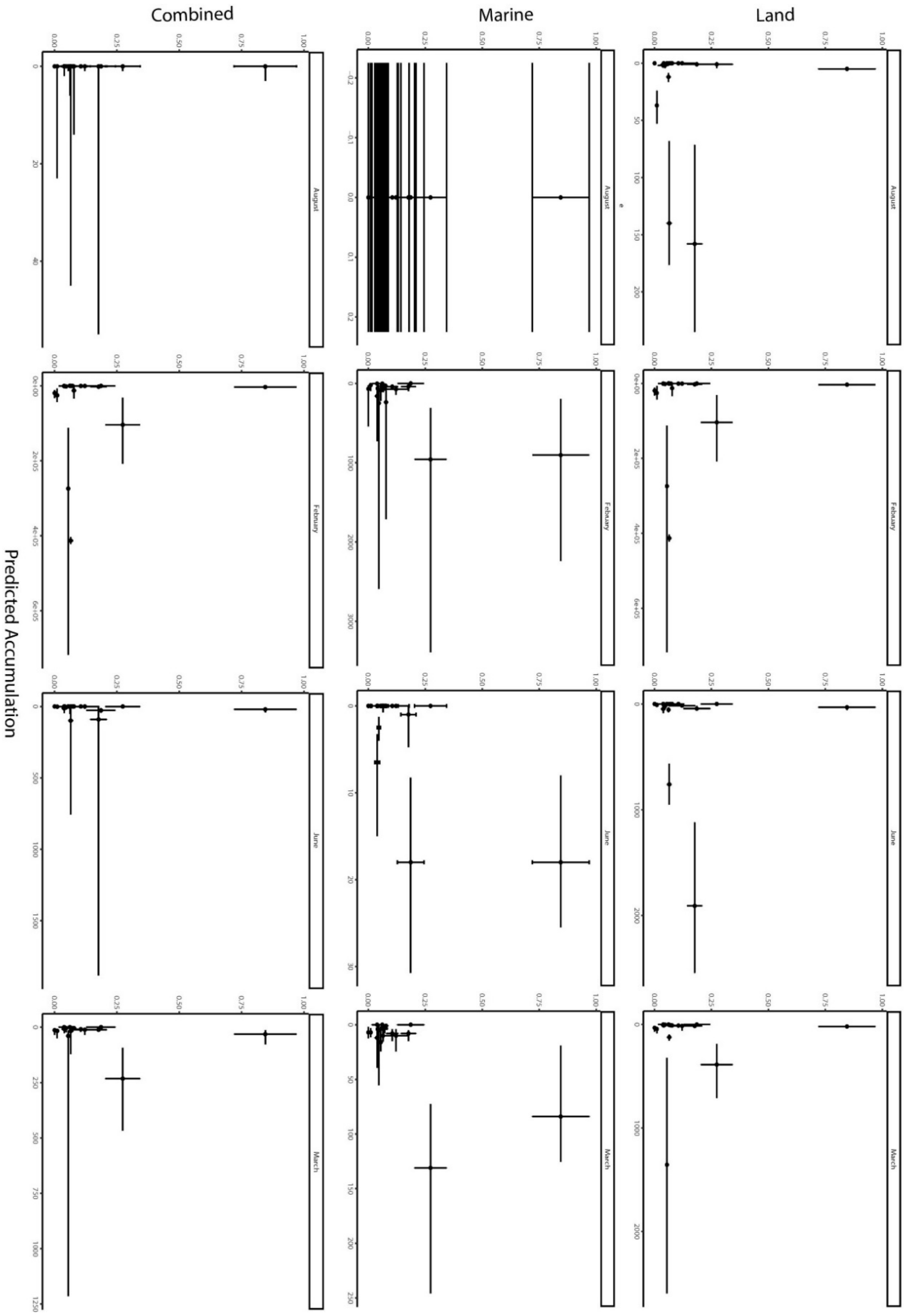
Xanthos, D. & Walker, T. R., 2017. International Policies to Reduce Plastic Marine Pollution from Single-Use Plastics (Plastic Bags and Microbeads): A Review. Marine Pollution Bulletin 118**,** 17-26.
Yoon, J. H., Kawano, S. & Igawa, S., 2010. Modeling of Marine Litter Drift and Beaching in the Japan Sea. Marine Pollution Bulletin 60**,** 448-63.
Zhang, H., 2017. Transport of Microplastics in Coastal Seas. Estuarine, Coastal and Shelf Science.
Zhao, S., Zhu, L., Wang, T. & Li, D., 2014. Suspended Microplastics in the Surface Water of the Yangtze Estuary System, China: First Observations on Occurrence, Distribution. Marine Pollution Bulletin 86**,** 562-568.

Appendix 1

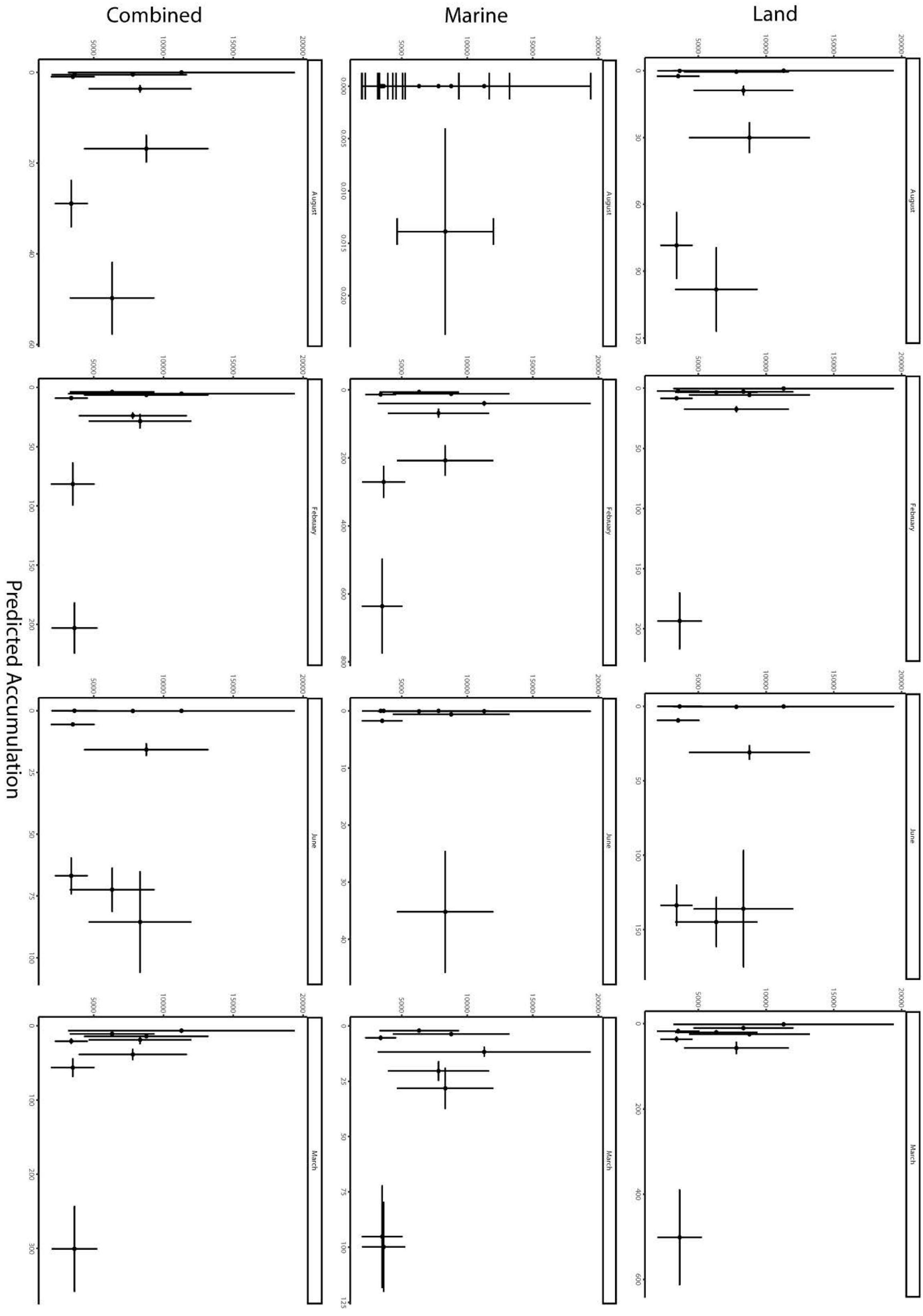
```
"""This script will count the number of particles of each type, for each
timestep, in a set of boxes defined in a file by the
user.
Producing a file with the number of particles in the box for every time step
for each boxID in the BoxLimits file
the box limits must be in decimal degrees in format
<SiteID>,<MaxLat>,<MinLat>,<MaxLon>,<MinLon> with no header.
written by Kay Critchell July 2017"""


import os, csv

def main():
    findFiles(r'D:\yourFileLocation', "yourLogFileName.txt",
"yourCoordsFile.csv") #define your top directory location, logfileName and
boxLimitsfile name

def findParticles(dirpath, fileName, boxLimits):
#initiate particle counts
    numberOfParticles = 0
    numberOfTag1 = 0
    numberOfTag0 = 0

    fileIn = str(dirpath + '\\' + fileName)
    print (fileIn)
    try:
        fIn = open(fileIn, "r")
        l = fIn.readline()
        while l != "":
            parts = l.split(" ")
            if float(parts[2]) == 2:
                if float(parts[3]) > float(boxLimits[4]) and float(parts[3]) <
float(boxLimits[3]) and float(parts[4]) > float(boxLimits[2]) and
float(parts[4]) < float(boxLimits[1]):
                    print(l)
                    numberOfParticles = numberOfParticles + 1
                    print(numberOfParticles)
                    if parts[1] == "0" or  parts[1] == "0.0\n":
                        numberOfTag0 = numberOfTag0 + 1
                    elif parts[1] == "1" or parts[1] == "1.0\n":
                        numberOfTag1 = numberOfTag1 + 1
            l = fIn.readline()

    except FileNotFoundError:
        print("file not found")

    print (boxLimits[0], numberOfParticles, numberOfTag0, numberOfTag1)
    return boxLimits[0], numberOfParticles, numberOfTag0, numberOfTag1



def findFiles(topDir, logname, boxLimitsFile):     #name of the log file that
will be saved in the same directory as this script
    # The top argument for walk: where you define your directory
    topdir = topDir
    # The extension to search for
    exten = '.txt'
    # What will be logged
```

```
    results = str()
    ignore = ['images', 'scripts', 'processed', "All"] # folders to ignore
that are in the directory tree

    limitsList = getCSVrows(boxLimitsFile)

    for limit in limitsList:
        for dirpath, dirnames, files in os.walk(topdir):
        # Remove directories in ignore list
        # directory names must match exactly!
            for idir in ignore:
                if idir in dirnames:
                    dirnames.remove(idir)
            print(dirnames)

            for name in files:
                if name.lower().endswith(exten) and name.lower() !=
logname.lower() and name.lower() != "triangles.txt":
                    siteID, numberOfParticles, numberWithTag0, numberWithTag1
= findParticles(dirpath, name, limit)
                    # Save to results string
                    results += '%s\t' % os.path.join(dirpath, name) + "%s\t"
%siteID + "%s\t" % numberOfParticles + " %s\t" % numberWithTag0 + "%s\n" %
numberWithTag1
                    # Write results to logfile when finished
        with open(logname, 'w') as logfile:
            logfile.write(results)


def getCSVrows(fileLoc):
    with  open(fileLoc, "r") as inFile:
        row_list = list(csv.reader(inFile))
    return(row_list)


main()
```

Appendix 2



A2.1 Figure: *Scatter plots showing the correlations for each scenario of the predicted and observed accumulation of microplastics.*



A2.2 Figure: *Scatter plots showing the correlations for each scenario of the predicted and observed accumulation of macroplastics.*

Appendix 3

A3.1 Table: Feeding regime for the chronic and acute exposure experiments

			Acute exposure		week 1		week 2		week 3		week 4		week 5		week 6	
clutch ID	Tank ID	% Diet as plastic	food (mg - l)	plastics (mg l-1)	food (mg - l)	plastics (mg l-1)	food (mg - l)	plastics (mg l-1)	food (mg - l)	plastics (mg l-1)	food (mg - l)	plastics (mg l-1)	food (mg - l)	plastics (mg l-1)	food (mg - l)	plastics (mg l-1)
A	15A	20	0.083	0.021	0.100	0.020	0.113	0.023	0.140	0.028	0.157	0.031	0.157	0.031	0.174	0.035
A	2A	20	0.080	0.020	0.100	0.020	0.113	0.023	0.138	0.028	0.146	0.029	0.146	0.029	0.162	0.032
A	14A	40	0.066	0.044	0.100	0.040	0.125	0.050	0.166	0.067	0.181	0.072	0.181	0.072	0.210	0.084
A	3A	40	0.069	0.046	0.100	0.040	0.128	0.050	0.165	0.066	0.163	0.065	0.163	0.065	0.225	0.090
A	13A	60	0.040	0.060	0.100	0.060	0.105	0.063	0.135	0.081	0.147	0.088	0.147	0.088	0.167	0.100
A	4A	60	0.041	0.062	0.100	0.060	0.110	0.065	0.141	0.085	0.151	0.091	0.151	0.091	0.173	0.104
A	12A	80	0.022	0.086	0.100	0.080	0.110	0.088	0.140	0.112	0.145	0.116	0.145	0.116	0.168	0.134
A	5A	80	0.018	0.071	0.100	0.080	0.090	0.070	0.116	0.092	0.123	0.098	0.123	0.098	0.140	0.112
A	11A	Control	0.104	0.000	0.100	0.000	0.113	0.000	0.130	0.000	0.134	0.000	0.134	0.000	0.152	0.000
A	1A	Control	0.098	0.000	0.100	0.000	0.108	0.000	0.132	0.000	0.146	0.000	0.146	0.000	0.171	0.000
B	19B	20	0.045	0.011	0.100	0.020	0.065	0.013	0.082	0.016	0.089	0.018	0.089	0.018	0.106	0.021
B	7B	20	0.059	0.015	0.100	0.020	0.080	0.015	0.100	0.020	0.106	0.021	0.106	0.021	0.125	0.025
B	18B	40	0.041	0.027	0.100	0.040	0.078	0.030	0.096	0.039	0.103	0.041	0.103	0.041	0.124	0.050
B	8B	40	0.035	0.023	0.100	0.040	0.063	0.025	0.070	0.028	0.076	0.030	0.076	0.030	0.093	0.037
B	17B	60	0.035	0.053	0.100	0.060	0.093	0.055	0.122	0.073	0.138	0.083	0.138	0.083	0.175	0.105
B	9B	60	0.030	0.045	0.100	0.060	0.083	0.048	0.099	0.059	0.106	0.063	0.106	0.063	0.127	0.076
B	10B	80	0.015	0.058	0.100	0.080	0.075	0.058	0.090	0.072	0.093	0.074	0.093	0.074	0.108	0.087
B	16B	80	0.013	0.052	0.100	0.080	0.068	0.055	0.086	0.069	0.088	0.071	0.088	0.071	0.107	0.085
B	20B	Control	0.091	0.000	0.100	0.000	0.110	0.000	0.137	0.000	0.149	0.000	0.149	0.000	0.185	0.000
B	6B	Control	0.062	0.000	0.100	0.000	0.070	0.000	0.087	0.000	0.094	0.000	0.094	0.000	0.110	0.000
C	2C	20	0.125	0.030	0.150	0.030	0.157	0.031	0.163	0.033	0.163	0.033	0.163	0.033	0.163	0.033
C	7C	20	0.118	0.028	0.138	0.028	0.145	0.029	0.154	0.031	0.158	0.032	0.163	0.033	0.163	0.033
C	3C	40	0.098	0.065	0.163	0.065	0.163	0.065	0.163	0.065	0.163	0.065	0.163	0.065	0.163	0.065
C	8C	40	0.090	0.059	0.148	0.058	0.148	0.059	0.158	0.063	0.164	0.065	0.163	0.065	0.163	0.065
C	4C	60	0.060	0.093	0.150	0.090	0.152	0.091	0.162	0.097	0.163	0.098	0.163	0.098	0.163	0.098
C	9C	60	0.060	0.090	0.143	0.085	0.146	0.088	0.154	0.093	0.162	0.097	0.163	0.098	0.163	0.098
C	10C	80	0.033	0.130	0.150	0.120	0.162	0.129	0.163	0.130	0.163	0.130	0.163	0.130	0.163	0.130
C	5C	80	0.030	0.118	0.135	0.108	0.144	0.115	0.154	0.123	0.163	0.130	0.163	0.130	0.163	0.130
C	1C	Control	0.145	0.000	0.140	0.000	0.145	0.000	0.149	0.000	0.156	0.000	0.162	0.000	0.163	0.000
C	6C	Control	0.140	0.000	0.135	0.000	0.142	0.000	0.148	0.000	0.153	0.000	0.162	0.000	0.163	0.000

A3.2 Table: *Feeding regime for the particle size experiment*

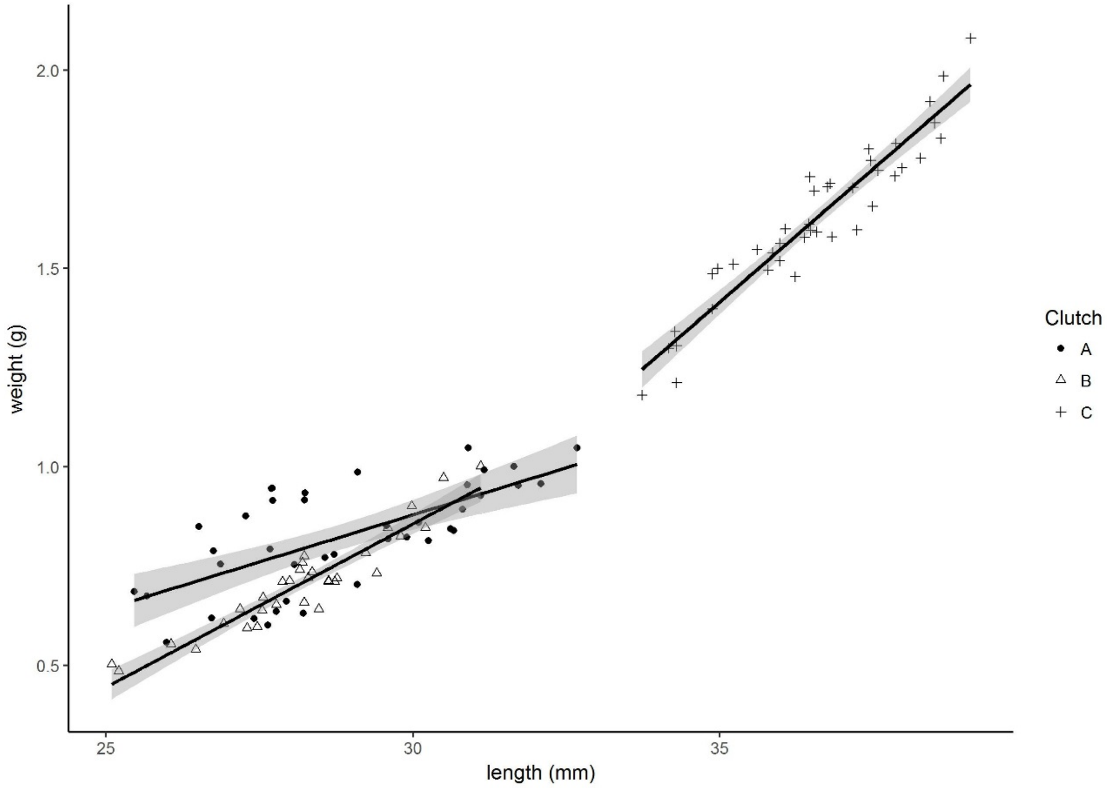
Tank Number	Fish Size Class	Plastic size class	Food (mg l^{-1})	plastics (mg l^{-1})
3	Small	Small	0.065	0.05
4	Large	Large	0.14	0.1
5	Small	Small	0.0775	0.06
6	Large	Large	0.1625	0.13
7	Large	Medium	0.15	0.12
8	Small	Medium	0.1	0.08
9	Small	Small	0.085	0.0675
10	Small	Large	0.1325	0.105
11	Small	Large	0.135	0.1075
12	Large	Large	0.1625	0.13
13	Large	Medium	0.1475	0.1175
14	Small	Medium	0.085	0.0675
15	Large	Small	0.1625	0.13
16	Large	Medium	0.1625	0.130
17	Large	Small	0.1625	0.130
18	Small	Medium	0.095	0.075
19	Large	Small	0.1625	0.130
20	Small	Large	0.11	0.088

The different clutches of juvenile *A. polyacanthus* had drastically different length to weight relationships. When we applied a linear model to the data the clutches were statistically different from each other therefore clutch was kept as a variable in subsequent analyses.

A3.11 Table: Table of ANOVA results with the response variable initial weight

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Initial Length	1	20.1960	20.1960	3632.957	< 2.2e-16
Clutch	2	0.2187	0.1093	19.668	5.440e-08
Initial Length : Clutch	2	0.3750	0.1875	33.727	4.617e-12
Residuals	106	0.5893	0.0056		

A3.12 Figure: Length weight relationships of the 3 clutches at the start of the experiment.



A3.13 Table: Coefficients of the three clutches

Clutch	Intercept	Slope
A	-0.54460085	0.04746250
B	-1.06841405	0.03484301
C	-2.73881573	0.08678282