Accepted Manuscript

Predicting the exposure of coastal species to plastic pollution in a complex island archipelago

Kay Critchell, Mark Hamann, Natalie Wildermann, Alana Grech

PII: S0269-7491(19)30148-4

DOI: https://doi.org/10.1016/j.envpol.2019.06.031

Reference: ENPO 12735

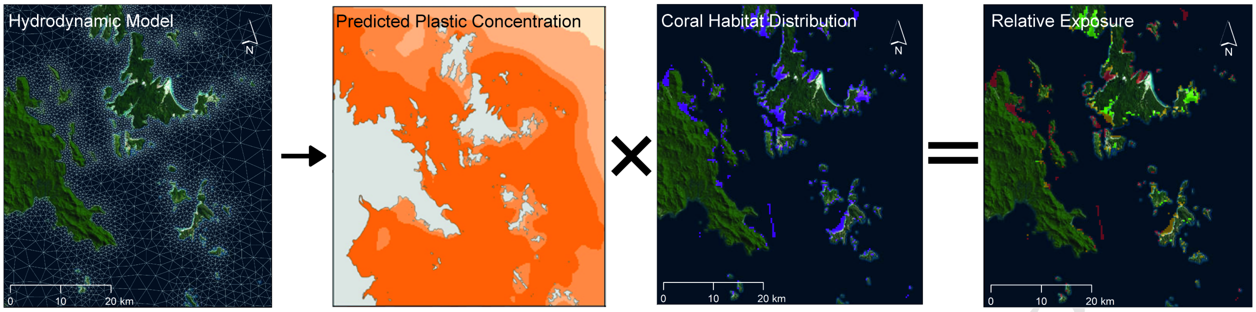
To appear in: *Environmental Pollution*

Received Date: 11 January 2019

Revised Date: 5 June 2019

Accepted Date: 8 June 2019

Please cite this article as: Critchell, K., Hamann, M., Wildermann, N., Grech, A., Predicting the exposure of coastal species to plastic pollution in a complex island archipelago, *Environmental Pollution* (2019), doi: https://doi.org/10.1016/j.envpol.2019.06.031.



Hydrodynamic Model
N
0
10
20 km
Predicted Plastic Concentration
Coral Habitat Distribution
N
0
10
20 km
Relative Exposure
N
0
10
20 km

Title: Predicting the exposure of coastal species to plastic pollution in a complex island archipelago

Authors: Kay Critchell [a,b*], Mark Hamann [a], Natalie Wildermann [a], and Alana Grech [c]

a. College of Science and Engineering and TropWATER, James Cook University, Townsville, Australia
b. Marine Spatial Ecology Lab, University of Queensland, St Lucia, Brisbane, Australia
c. Australian Research Council Centre of Excellence for Coral Reef Studies, James Cook University, Townsville, Australia

*Corresponding author:

E-mail address: kay.critchell@my.jcu.edu.au

Phone number: +61 420 597 025

Current postal address: The Science Place, James Cook University, Townsville, Queensland, Australia, 4811

Acknowledgments

We would like to acknowledge the expertise of Dr Jonathan Lambrechts, (of the Université Catholique de Louvain, Belgium) for the mesh used for the models and technical support for the model runs. The authors also wish to thank the effort of the three anonymous reviewers, their input vastly improved this manuscript. The turtle tracking was initially funded by the Mackay & District Turtle Watch Association, WWF-Australia, Queensland Department of Environment and Science, and Gladstone Ports Corporation's Ecosystem Research and Monitoring Program. This research did not receive any specific grant from funding agencies in the public, commercial, or not-for-profit sectors.

Abstract

Plastic pollution in the marine environment is a pervasive and increasing threat to global biodiversity. Prioritising management actions that target marine plastic pollution require spatial information on the dispersal and settlement of plastics from both local and external sources. However, there is a mismatch between the scale of most plastic dispersal studies (regional, national and global) and the scale relevant to management action (local). We use a fine-resolution hydrodynamic model to predict the potential exposure of coastal habitats and species (mangroves, coral reefs and marine turtles) to plastic pollution at the local scale of a management region (the 1,700km^2 Whitsunday Islands, Queensland, Australia). We assessed the potential exposure of mangroves, coral reefs and marine turtles to plastics during the two dominant wind conditions of the region; the trade wind and monsoon wind seasons. We found that in the trade wind season (April to September) all habitats and species had lower exposure than during the monsoon wind season (October to March). In both wind seasons we found a small proportion of coral reef habitat and large area of turtle habitat were in high potential exposure categories. Unlike coral reefs or marine turtles, mangroves had consistent hotspots of high exposure across wind seasons. Local scale management requires data at fine resolution to capture the variability that occurs at this scale. The outputs of our study can inform the development of conservation resources and local scale management action.

Keywords: dispersal modelling, macroplastics, microplastics, coral reefs, marine turtles, mangroves

1. Introduction

The allocation of limited conservation resources is often dependent on information on the spatial distributions of assets and threatening processes (Ban 2009; Halpern et al. 2015). Therefore, spatially-explicit modelling is often used by scientists, and managers to predict the distributions of species and threats (e.g. Grech et al. 2011; Grech and Marsh 2008; Halpern et al. 2015; Halpern et al. 2008; McPherson et al. 2008). Modelling is especially useful in the marine environment where data is 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 used to determine distribution of wide-ranging marine species such as marine turtles and sharks (McKinney et al. 2012; Wildermann et al. 2017). Hydrodynamic modelling is also used to predict how threats such as chemical pollutants are distributed and diluted away from the source (Cucco and Daniel 2016; Li et al. 2000).

Plastic pollution is a threat of concern to the marine environment. The negative consequences of exposure to plastic is only beginning to be understood for a variety of species (Worm et al. 2017). For example, marine turtles ingest small plastic particles, causing disruption to their gastrointestinal tract (Colferai et al. 2017; Di Bello et al. 2013), and the plastic particles have the potential to leach adsorbed chemicals, negatively impacting animal health (Andrady 2011; Rochman et al. 2014). Sensitive habitats, such as mangroves and coral reefs, can be damaged by scouring or smothering by larger plastic items (e.g. Donohue et al. 2001; Smith 2012; Uneputty and Evans 1997). The effects of plastic exposure to marine turtles, coral reefs and mangroves, both evidenced and speculated are summarised in Table 1. However, the spatial location of where these interactions occur, and the frequency of interactions, remains poorly understood (Nelms et al. 2016; Titmus and Hyrenbach 2011), especially on small scales. Hydrodynamic 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 course spatial resolution. Such a broad spatial resolution is often inadequate to inform management of pollutants, such as plastics, at a fine spatial scale (e.g. the Whitsundays Special Management Area [Great Barrier Reef Marine Park Authority]: ~1700 km^2; Figure 1). Management of plastics in the current political climate is often conducted at the small scales of local management authorities. These authorities have limited resources and scope to manage the whole issue, and therefore must strategize to their manageable waste, i.e. waste created in the management region, that is affecting their management region.

Predictions on where plastics are accumulating are important for forecasting the location of potential interactions with environmental features. Hydrodynamic modelling has been used previously to estimate the concentrations and distribution patterns of plastics over large areas, such as global oceans and seas (Carson et al. 2013; Eriksen et al. 2014; Lebreton et al. 2012; Maximenko et al. 2015). However, it is also important to understand the distribution of plastics in the coastal zone and at smaller jurisdictional scales (Critchell and Lambrechts 2016) because it is at this scale that interactions between plastics, coastal species and habitats occur, and it is a management scale for which intervention action can be more readily implemented.

The goal of this study is to predict the spatial distribution of exposure of coral reefs, mangrove habitats, and foraging flatback marine turtles to plastic pollution within the management control of a small jurisdiction, the Whitsunday region, Queensland, Australia (Figure 1). To achieve this, we used hydrodynamic modelling to estimate accumulation areas of plastics. From the accumulation estimates, we created exposure categories, which were then compared with known distributions of coral reefs, mangroves and turtles. The outputs of our approach provide a tool to improve the management of plastic pollution in the coastal zone.

2. Methods

2.1 Study area and species

Our study location was the Whitsunday region of the Great Barrier Reef World Heritage Area (GBRWHA), on the central coast of Queensland, Australia (Figure 1). The region has an average depth of approximately 30 metres, and is punctuated with 77 islands and reefs. The Whitsunday region is towards the southerly extreme of the tropics (-20 degrees latitude), and is dominated by the monsoonal and south-easterly trade wind circulations. During the monsoonal austral summer months, the region receives run-off from three mainland catchments predominantly of agricultural (grazing and crops) land-use (Figure 1). The region attracts a large transient tourist population (~700,000 visitors per year; Tourism Research Australia, 2018).

Coral reefs are one of the key environmental values of the GBRWHA. The health of coral reefs is important, in the Whitsunday region in particularly, due to the high tourism value to the region. Mangrove habitats serve as important nursery grounds for many commercially important species, as well as providing multiple ecosystem services in the coastal zone.

Flatback turtles (*Natator depressus*), endemic to Australia, are listed as "Vulnerable" under the EPBC Act (1999) and are an important value of the GBRWHA. There is growing evidence that marine turtles are at risk from plastic exposure (e.g. Schuyler et al. 2012, 2014), and plastic debris is recognised as a threat to flatback turtles residing in the GBRWHA and elsewhere in Australia (summarised in Table 1). Approximately 19 turtles (species not available) per year 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 (StrandNet - Department of Environment and Heritage Protection, 2017). Four of the five flatback turtles that stranded in the Whitsundays between 2013 and 2017 could be necropsied and they had microplastic blockage in their gastrointestinal tract (unpublished data), which is supported by Duncan et al. (2018) who found every turtle examined had microplastics in their digestive tract.

Spatial data on corals and mangroves were obtained from Geosciences Australia and the habitat use data for flatback turtles was extracted from Wildermann et al. (in review). For full descriptions, see the online appendix.

2.2 Hydrodynamic modelling and dispersal simulations

We used the Second-generation Louvain-la-Neuve Ice-ocean Model (SLIM; full documentation and code available www.climate.be/slim) described in Critchell and Lambrechts (2016) to model the dispersal of plastic pollution in the Whitsunday region (Figure 1). The SLIM is a depth-averaged finite element hydrodynamic and advection-dispersion model. Although a three-dimensional approach would be beneficial, the depth-averaged approach is justified in Critchell et al. (2015) where it is shown the currents are similar though the water column in this well mixed region. The hydrodynamic part of the model is forced by physical data on winds, tides and inflow from the Coral Sea. The advection-dispersion part of the model uses the velocity field created by the hydrodynamic model and uses the Lagrangian scheme of particle movement to simulate pathways in the study area. The particles (which can be thought of as simulated plastics) also move with plastic-specific properties such as degradation from macroplastics to microplastics, wind drift, beaching, and resuspension/re-floating from the coastline (see below for details). Data inputs to both parts of the model included recorded sea surface elevation from Shute Harbour (Australian Bureau of Meteorology), and wind data collected at Shute Harbour (Australian Bureau of Meteorology, station number 33106) from June 1 2013 to May 31 2014. This enabled the comparison of plastic pollution distribution during the southeast trade wind season (April - September) and the more wind-variable northerly season (hereafter, referred to as "monsoon wind season"; October - March) for this year of data. It is important to compare these seasons as the effect of wind strongly influences the movements of marine plastics (Critchell et al. 2015; Critchell and Lambrechts 2016), and the two wind seasons capture the maximum variability during one year. The years 2013 and 2014 provide good examples of the typical conditions during the two wind seasons, showing typical wind distribution patterns for the seasons (Figure 2). We imposed a constant wind shadow in the lee of islands of 2500 m. The model was forced with a standardised forcing from the Coral Sea. The hydrodynamic model used in our analysis has been validated in previous studies and found to provide an acceptable representation of water movements (Lambrechts, et al., 2008).

To seed the dispersal model, (i.e. choose the starting locations for the particles) we used the most likely sources of manageable plastic pollution for the study area (Figure 1 and Table 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 (Critchell et al. 2015; Moore et al. 2011; Schmidt et al. 2017). We 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. We included Hamilton Island as a source of both macro- (tourist and resort-based litter) and microplastics from the water treatment plant (e.g. from cosmetic products). Offshore sources were evenly spaced along the offshore commercial shipping lane. We 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 precise seeding location is not obvious. As the focus of this study is a small management region, we did not include external coastal sources in this model. This was to ensure the focus remained on sources within the management authority's control. The importance of a source was represented in the model by increasing the number of seeding locations, as 250,000 particles were released at each source point (see Table 2 for justification for each seeding location). Particles were released instantaneously, at the same phase of the tide for each scenario.

The dispersion model uses the velocities derived by the hydrodynamic model to move particles according to the Lagrangian dispersion scheme. Within this, we also added plastic-specific parameters, such as wind drift, resuspension, and degradation rate (Critchell and Lambrechts, 2016). To predict the distribution of plastics in the Whitsundays region, we used the best estimation of these parameters, informed by available literature. The wind is used in the hydrodynamic model to influence the water velocities, however buoyant objects are also directly influenced by the wind, through wind drift. We set the wind drift to 2% of the wind velocity for simulated macroplastic particles and zero for microplastics, as we assume they did not break the surface and so are not directly influenced by the wind. Particles are considered on the coastline (beached) if the particle is moved past a coastal boundary (e.g. by wind). The resuspension (or re-floating) was set to a probability of 0.2 per day for both macro- and micro- beached particles. The probability was set to 0.2 per day to represent the likelihood of resuspension following the tidal cycle over the day. We assume the resuspension is mainly wave driven, therefore the resuspension is turned off in the areas designated as being in a wind shadow, as wind driven waves would be absent in these areas, and this region is not influenced by swell waves due to blocking by the off-shore reefs of the Great Barrier Reef. The wind shadow was set to the same value as for the hydrodynamics model (2500 m). The rate of settling is set as a probability, 0.002 per time step (300 seconds) for macroplastics and 0.02 per time step for microplastics, as microplastics are more likely to become bio-fouled that will cause sinking, or to be flocculated into marine snow. The rate at which macroplastics "degrade" into microplastics is $1x10^{-6}$ per day for particles in suspension and $1x10^{-5}$ per day 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 decomposition is thought to be at the scale of months to years and the rate used here may be still too fast, providing a worst case scenario of microplastics production. We ran one simulation for each wind season. The trade wind season simulation began on 1st June 2014, and the monsoon wind season simulation started 1st Feb 2014 and ran for 45 days (Figure 2).

2.3 Plastic exposure layers

The output particle locations from the two simulations were imported to the geographic information system, ArcGIS 10.2 (ESRI). The outputs were used to create macroplastic and microplastic exposure layers for each wind season (trade wind and monsoon), and each particle state (suspended particles, beached particles and settled particles), resulting in 12 exposure layers (see online Appendix 2).

To delineate the distribution of settled particles, we used the accumulative locations of settled particles throughout the simulation. We created a density distribution, with the Kernel Density function of ArcGIS based on the model outputs for macro- and microplastics. The result is a surface where each cell value represents the density of particles.

To delineate the distribution of suspended particles, we created a density distribution with the Kernel Density function of ArcGIS as for the settled particles. However, as each model output for suspended particles is the current location of the particles, we based the model outputs for macro- and microplastic on three outputs evenly spaced across the simulation, on day 15, 30 and 45, to capture variability in dispersal throughout the simulation length. We then used the mean particle density of these outputs to create the exposure layers for suspended macro- and microplastics for each wind season.

To delineate the distribution of beached particles we used the spatial join function in ArcGIS to join the simulated particle locations to their closest coastline section for each of the model outputs. We calculated the mean particle density of the outputs of days 15, 30 and 45 to create an exposure layer of the particle density of each 100 m section of coastline, for each wind season and plastic type (macro- and micro-). To compare the beached plastic exposure layers with the mangrove habitat, we joined the polyline of mangrove presence (see online appendix 1 for habitat distribution data) to the SLIM coastline in a binary format (zero = absent, one = present). We then multiplied the mean particle density by the habitat presence to assess the relative exposure of each 100 m section of mangrove habitat.

As the raw number of particles in a cell is dependent on the initial number seeded (true value unknown), the layers were categorised (binned) into four “Relative Exposure Categories” (RE categories) based on the number of particles predicted within each cell. The RE categories were: Nil (where plastics were not predicted to be present), low, medium, and high, see online Appendix 2 for details. The layers of each particle type (macro- and micro-) and state (beached, suspended and settled) had different particle density frequency distributions, so it was necessary to develop these categories individually, while retaining the ability to directly compare concentrations in the trade vs. monsoon wind layers. For example, suspended macroplastics in the trade wind season, and suspended macroplastics in the monsoon wind season must be comparable. The breaks used for the categories were based on the quantile distribution, which bins data into classes with an even number of grid cells within each class. There is insufficient data on the thresholds of concentrations that cause harm to habitats and species, therefore at this stage, only relative measures of exposure and risk were able to be assessed.

The response of turtles, reefs and mangroves to plastics in the three states (i.e. beached, suspended and settled) differs (see Table 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 (see Table 1), and mangroves are a coastal habitat and therefore more likely to be affected by plastics that are pushed onto the coastline (beached). We matched turtles, reefs and mangroves to their relevant exposure layer when conducting the exposure analysis (Table 1).

3. Results

The trade wind and monsoon wind seasons resulted in different spatial accumulation patterns of, settled, suspended and beached macro- and microplastics (see online Appendix 2). The trade wind season moved particles representing macro- and microplastics into the large south-east facing bay at the southern end of the Whitsunday region (Repulse Bay, Figure 1). In comparison, the monsoon wind season moved particles into the smaller, more complex bays in the north of the study region (e.g. Double Bay, see Figure 1). The macroplastics accumulate close to the northeast facing coastlines during the monsoon wind season, while microplastics had more even distribution. In the trade wind season, both plastic types had discrete locations of high accumulation (Figure 3). Microplastics accumulated on the beaches of the Lindeman group (southern end of the study region, Figure 1) in the trade wind season, however there is no equivalent accumulation for the macroplastics.

The movement patterns of particles between seasons also differed (Figure 3). During the trade wind season, the simulated microplastics moved a median of 54 km to the north (35 km and 64 km; 25th and 75th percentiles respectively), whereas macroplastics moved a median 447 km north after 45 days of simulation (269 km and 559 km; 25th and 75th percentiles respectively). A smaller proportion of simulated microplastic and macroplastic moved south of their original source location during the trade wind season simulation, 21% and <1% respectively. By contrast, during the monsoon wind season the simulated microplastics moved a median of 64 km (54 km and 75 km; 25th and 75th percentiles respectively), and macroplastics moved 76 km (60 km and 77 km; 25th and 75th percentiles respectively). A large amount of microplastics and macroplastic particles (60%, and 46%, respectively) moved south of their source location during the monsoon wind simulation.

Coral reefs had the largest proportion of habitat area constantly in the 'nil' RE category, a mean of 58% of habitat across season and plastics type (Figure 4). Coral reefs had extremely low potential exposure during the trade wind season, with 95% and 99% of the habitat in low and nil RE categories, for macroplastic and microplastics, respectively. Conversely, during the monsoon winds season 32% of the habitat was in the medium or high RE category for macroplastics and 28% for microplastics (Figure 4). Only one cell (equivalent to 0.11 km^2) was consistently in the high RE category across wind season for microplastics and none for macroplastics (Figure 5). The single cell in high exposure category is in Repulse Bay near the mouth of the Proserpine River a major source location of marine plastics (Figure 1).

Similarly, the flatback turtle home ranges have a large area in the medium and high RE category in the monsoon wind season, compared with a small proportion during the trade wind season (for microplastics, 51% and 2%, respectively). Unlike the coral reef habitat however, there is a clear difference in potential 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% for microplastics (Figure 3 & 4). Only 20 km^2 of flatback turtle home range (<1% of the total area) was consistently in the medium or high microplastic RE category between seasons, while only two km^2 of the area was consistently in medium and none in the high RE category for macroplastics (Figure 5).

The mangrove habitat had complex exposure patterns. Mangroves had the smallest proportion of its range in the nil RE category compared to marine turtles and coral reefs (mean 20% mangrove area across season and plastic type; Figure 4). Unlike the coral reef habitat and the turtle home range, the proportions of mangrove habitat in each RE category are 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 regardless of season, suggesting there are hotspots of exposure that are consistent in time and space (e.g. much of the mangrove habitat in Pioneer Bay, surrounding Airlie Beach; Figure 5).

4. Discussion

We developed and demonstrated a method to predict, spatially, the exposure of coral reefs, mangroves, and foraging flatback marine turtles to plastic pollution in a complex, coastal environment. Weather condition was the primary driver of potential exposure levels, which differed during each wind season (trade wind and monsoon) and for each plastic type (macro- and micro-). During the trade wind season plastics were moved out of the study area by local wind and water circulation patterns, reducing the potential interaction between plastics and coral reefs, mangroves and turtles in the region. 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 in the lee side of islands and protected bays (Hopley 1982; Kennedy and Woodroffe 2002), as pervasive exposure to wind generated waves can damage the coral structure and restrict growth. We found that during the monsoon wind season, sheltered reef habitats were more exposed to plastics than during the trade wind season because plastics accumulate in the naturally protected habitats.

A large proportion of mangrove habitat had a relatively high exposure to plastic pollution in both wind seasons and for both plastic types (for maps, see online Appendix 3). However, the risk posed by macro- and microplastics is likely to be different. For example, a single large object such as plastic sheeting or fishing gear can damage a comparatively large area of mangrove habitat (Goldberg 1997; Uneputty and Evans 1997). Whereas, the concentration of microplastics necessary to cause an effect is likely to be higher, for example, sediment permeability in beach sediments is significantly affected at a concentration of approximately 16% of the sediment as plastic by weight (Carson et al. 2011). Therefore, the concentration of macroplastics (represented by the density of plastics per area of coastline in this study) required to have a negative consequence on mangrove habitat could be much smaller than for microplastics. Even though a similar area of mangroves 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, coral reefs had a larger area exposed to the highest exposure category of microplastics than it did to macroplastics (for maps, see online Appendix 3), with one cell in the high exposure category consistently in both seasons. This cell is one of only two in Repulse Bay, at the mouth of the Proserpine River. It is possible that the cell is consistently exposed due to its proximity to the modelled source of plastics, highlighting the importance of source reduction. Microplastics threaten reef habitats as they can be ingested by reef animals, including reef-building corals (Hall et al. 2015). However, this impact would be at a polyp or individual colony level, so it may be that macroplastics have a larger and more immediate consequence to reef habitats as a whole; scouring and smothering by plastics can affect the reef structure by damaging corals on a larger scale than individual coral polyps or colonies. For example, Chiappone et al. (2005) found up to 11 individual reef organisms damaged by a single piece of derelict fishing gear in the Florida Keys. There is also emerging evidence that macroplastic presence on reefs increases prevalence of coral disease (Lamb et al. 2018). Macroplastic pollution adds an additional threat to reef habitats in the region and pervasive exposure to plastics could affect natural recovery after environmental disturbance (Hughes et al. 2015). Some types of debris on coral reefs are deposited *in situ* (e.g. fishing line; Chiappone et al. 2005), however the types and accumulation of debris on the seafloor in the GBRWHA is likely to be variable (Bauer-Civiello et al. 2018). The variety of debris on reefs highlights the need to modify modelling parameters to represent debris types that are most threatening for the receptor of interest. For example, to understand the true plastic exposure on reefs, it would therefore be necessary to include these *in situ* sources alongside the remote sources we have considered here. It would also be necessary to modify the drift parameters appropriately to represent these types of debris, for example fishing net that is drifting while in contact with the sea floor would have drastically different processes acting on it than the buoyant litter modelled here.

We found that large areas of one of the most important foraging areas for flatback turtles in the Great Barrier Reef (GBR) (Wildermann et al. *in review*) could be exposed to high levels of plastic pollution during the monsoon season. Entanglement in derelict fishing gear 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). A far larger, and largely unquantified, issue is ingestion (Nelms et al. 2016; Vegter et al. 2014). The inability of marine turtles to regurgitate means that exploratory bites tend to be swallowed and thus plastics get ingested regularly (Schuyler et al. 2012). As 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). Given the high potential exposure to microplastic in the monsoon season, the chances of encountering and ingesting plastic particles could be high. There is growing evidence of the transfer of harmful chemicals from plastic to the animals that consume them (Andrady 2011; Gall and Thompson 2015) potentially having sub-lethal impacts on the animal. Kelly et al. (2008) and Stewart et al. (2011) found that PCBs (a common plastic-associated compound) are transferred from female turtles to their eggs and affect the growth and development of the turtle hatchling. Comprehensive risk assessments that consider consequence in addition to exposure are unable to be conducted without estimates of the concentrations of plastics in the environment, or the amount of plastics it would be necessary to ingest, for sub-lethal affects to develop.

Despite turtles often being used as flagship species to highlight the problems of plastic pollution, there is limited information about their actual exposure to plastic pollution and the degree to which populations or species are impacted. A global risk assessment by Schuyler et al. (2016) assessed the hotspots of plastic ingestion by marine turtles, however the broad spatial scale (1^0 x 1^0) is inappropriate for local governance and intervention. Wilcox et al. (2013) developed a model using known ghost net locations combined with estimated turtle abundance and known turtle entanglement rates, to model the risk of turtles to ghost fishing at a scale relevant to the problem. The spatial scale used by Wilcox et al. (2013) was at a resolution of 5 x 5 degrees, much broader than the resolution of the model presented in this study. A coarse spatial resolution is appropriate for modelling across a large geographic range, however a broad scale model is insufficient for addressing local scale management questions. The fine-scale approach presented in our study enables a local-scale 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 exposure is greatest. 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 (Nelms et al. 2016). This study represents the first time the degree to which an important foraging habitat for an Australian marine turtle species could be exposed to microplastic pollution has been quantified.

In this study we use a resolution of 1 km^2 and ~0.1 km^2 enabling our results to be used to inform fine-scale decision making on waste and marine debris management in the Whitsunday region. This is an example of institutional fit, where management arrangements match “the defining features of the problems they address” (Young 2008). Our fine-scale approach can be used for targeting debris removal activities on specific beaches/areas of accumulation where impact may be largest. For example, overlaying these data with tourism visitation may trigger certain reefs to be targets for clean-up activities. The details of the sources may trigger targeted interventions on land to prevent an asset being exposed to plastic pollution. We acknowledge that the approach presented in this study has limitations. Most notably, the plastic specific dispersal modelling underlying the exposure analysis has not, at this stage, been validated by field data. A ground-truthing step would allow us to assess the uncertainty associated with the exposure analysis, and accurate abundance estimates would inform interaction rates. Ground-truthing could be conducted through field surveys of debris in these habitats, however, this is field intensive task. To obtain statistical significance in a system with high natural variability in space and time, a large number of samples from a large number of locations, under many environmental conditions are needed. The task quickly becomes logistically impossible with the current techniques of sample processing for microplastics. While we predict a large difference in the exposure between the two wind seasons, it is possible that plastics from external sources (e.g. cities to the south) are imported into the study area during the trade wind season, further reducing our ability to ground-truth the dispersal model focused on local supply with field data. It is likely that plastics in our study area, as with many parts of the Australian coastline, are predominantly from local sources (Hardesty et al. 2017). Even with these limitations, our study advances the prediction of exposure to plastic pollution on a local, management-relevant scales by using a high resolution hydrodynamic model. Our approach is readily scalable to larger or smaller jurisdictions and other habitats or taxa where there is existing information on hydrodynamics and species distributions. As the evidence on impacts of plastic pollution on habitats and species becomes more certain, future studies will be able to conduct ecological risk assessments that incorporate consequence of exposure levels on the receiving environment.

Conclusions

In this study, we used hydrodynamic models to predict exposure from plastic pollution for a multiple habitats and species of conservation concern. We found that in the monsoon wind season the habitats and species were highly exposed to plastic pollution, with few consistencies in locations of accumulation across season. The exception was for mangrove habitats, which had areas of consistently high exposure across wind seasons. The exposure data presented can be used in the prioritisation of conservation resources, from debris removal programs to locating offset initiatives. The framework we have used could also be refined to map the exposure of a particular plastic class, or types of objects that are known to affect a particular species or habitat e.g. fishing line on reef structures.

References

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

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.

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.

Bauer-Civiello, A., Loder, J., Hamann, M., 2018. Using citizen science data to assess the difference in marine debris loads on reefs in Queensland, Australia. Marine Pollution Bulletin 135, 458-465.

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, In Conservation Technical and Data Report 2010. pp. 1-124.

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.

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., Galloway, T., Thompson, R., 2007. Microplastic—an emerging contaminant of potential concern? Integrated Environmental Assessment and Management 3, 559-561.

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.

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.

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.

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.

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.

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.

Cucco, A., Daniel, P., 2016. Numerical Modeling of Oil Pollution in the Western Mediterranean Sea, pp. 1-20. Springer Berlin Heidelberg, Berlin, Heidelberg.

de Carvalho-Souza, G.F., Llope, M., Tinôco, M.S., Medeiros, D.V., Maia-Nogueira, R., Sampaio, C.L.S., 2018. Marine litter disrupts ecological processes in reef systems. Marine Pollution Bulletin 133, 464-471.

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.

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.

Duncan, E.M., Broderick, A.C., Fuller, W.J., Galloway, T.S., Godfrey, M.H., Hamann, M., Limpus, C.J., Lindeque, P.K., Mayes, A.G., Omeyer, L.C.M., Santillo, D., Snape, R.T.E., Godley, B.J., 2018. Microplastic ingestion ubiquitous in marine turtles. Global Change Biology 0.

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.

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.

Gall, S.C., Thompson, R.C., 2015. The impact of debris on marine life. Marine Pollution Bulletin 92, 170-179.

Goldberg, E.D., 1997. Plasticizing the Seafloor: An Overview. Environmental Technology 18, 195-201.

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.

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.

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.

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., 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., 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.

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.

Hopley, D., 1982. The geomorphology of the Great Barrier Reef : quaternary development of coral reefs. Wiley.

Hughes, T.P., Day, J.C., Brodie, J., 2015. Securing the future of the Great Barrier Reef. Nature Clim. Change 5, 508-511.

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.

Kelly, S.M., Eisenreich, K.M., Baker, J.E., Rowe, C.L., 2008. Accumulation and maternal transfer of polychlorinated biphenyls in snapping turtles of the upper Hudson River, New York, USA. Environmental Toxicology and Chemistry 27, 2565-2574.

Kennedy, D.M., Woodroffe, C.D., 2002. Fringing reef growth and morphology: a review. Earth-Science Reviews 57, 255-277.

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.

Lebreton, L., Greer, S., Borrero, J., 2012. Numerical modelling of floating debris in the world's oceans. Marine Pollution Bulletin 64, 653-661.

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.

Maximenko, N., MacFadyen, A., Kamachi, M., 2015. Modeling the drift of marine debris generated by the 2011 tsunami in Japan. PICES Press 23.

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.

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.

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.

Parga, M.L., 2012. Hooks and sea turtles: a veterinarian's perspective. Bulletin of Marine Science 88, 731-741.

Rochman, C.M., Hoh, E., Kurobe, T., Teh, S.J., 2013. Ingested plastic transfers hazardous chemicals to fish and induces hepatic stress. Sci Rep 3, 3263.

Rochman, C.M., Lewison, R.L., Eriksen, M., Allen, H., Cook, A.-M., Teh, S.J., 2014. Polybrominated diphenyl ethers (PBDEs) in fish tissue may be an indicator of plastic contamination in marine habitats. Science of The Total Environment 476-477, 622-633.

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.

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.

Stewart, K.R., Keller, J.M., Templeton, R., Kucklick, J.R., Johnson, C., 2011. Monitoring persistent organic pollutants in leatherback turtles (Dermochelys coriacea) confirms maternal transfer. Marine Pollution Bulletin 62, 1396-1409.

StrandNet, Queensland Department of Environment and Science Marine Wildlife Stranding Database, https://www.ehp.qld.gov.au/wildlife/caring-for-wildlife/strandnet-reports.html

Titmus, A.J., David Hyrenbach, 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.

Tourism Research Australia, 2018, Local government area profiles, 2017 Whitsunday, Queensland, Australian Government, Austrade, https://www.tra.gov.au/Regional/local-government-area-profiles

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.

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.

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.

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, n/a-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., 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.

Wildermann, N., Shimada, T., Barnett, A., Bell, I., Hof, C., Limpus, C., Fuentes, M., Hamann, M., in review. Dynamic migratory and foraging strategies of flatback turtles. Marine Ecology Progress Series.

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.

Young, O.R., 2008. The architecture of global environmental governance: bringing science to bear on policy. Global Environmental Politics 8, 14-32.

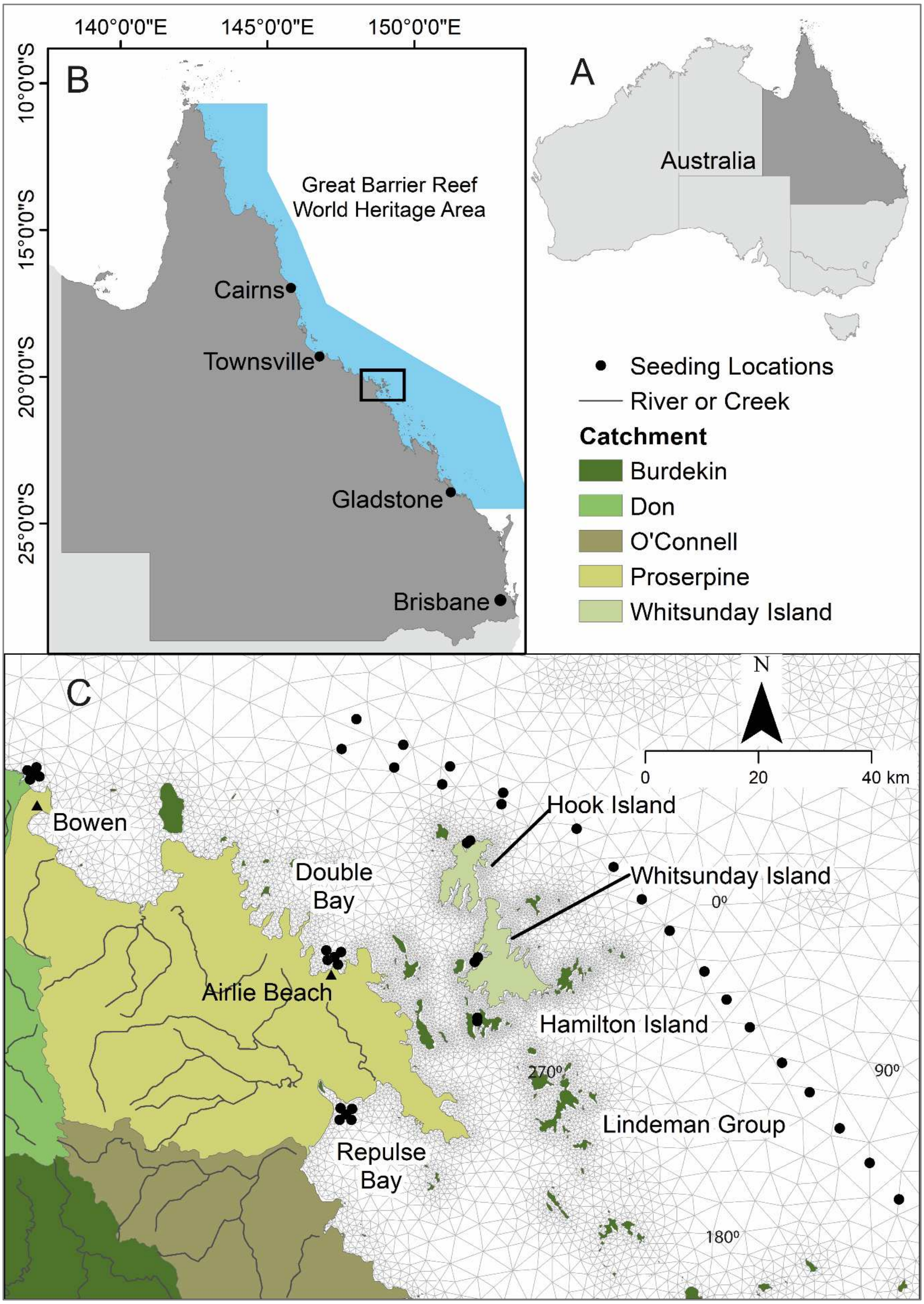


Figure 1: (A) shows Australia with the state of Queensland in dark grey; (B) the extend of the Great Barrier Reef World Heritage Area off the coast of Queensland (blue shaded area), and the location of the Whitsundays region as the black box; (C) the SLIM mesh as grey geometric lines and the placement of the hydrodynamic simulation seeding locations, shown as black circles. The river catchments are shown in green hues, with streams and rivers shown in dark grey.

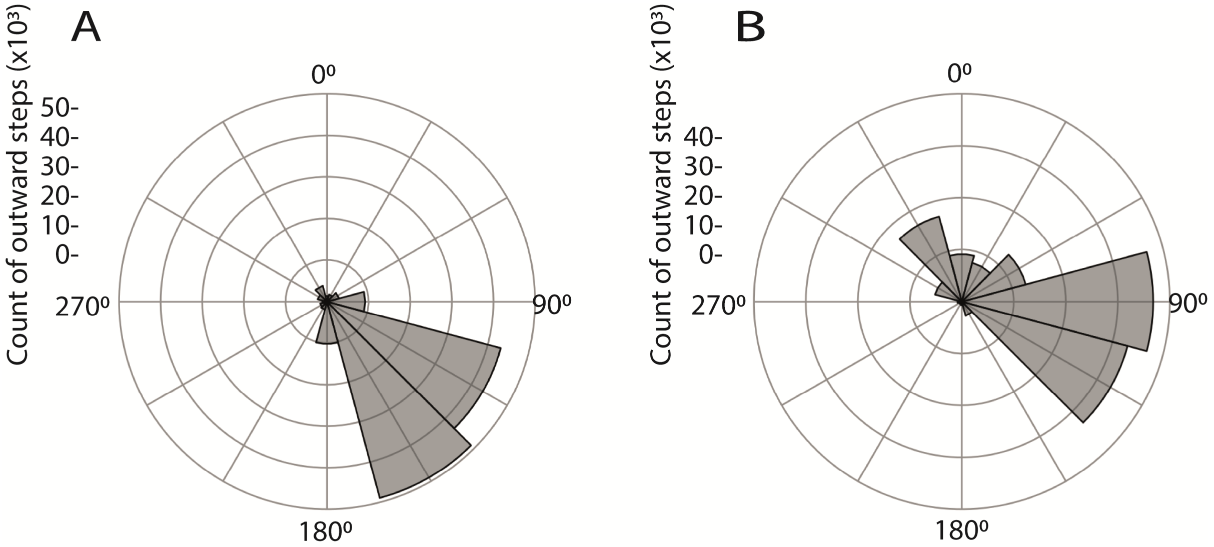


Figure 2: Wind rose of the wind data used in the hydrodynamic modelling used to create the exposure layers for each season, A) south-east trade wind season and B) monsoon wind season. Wind data from 10 min wind records at Shute Harbour weather station (Australian Bureau of Meteorology station number 33106).

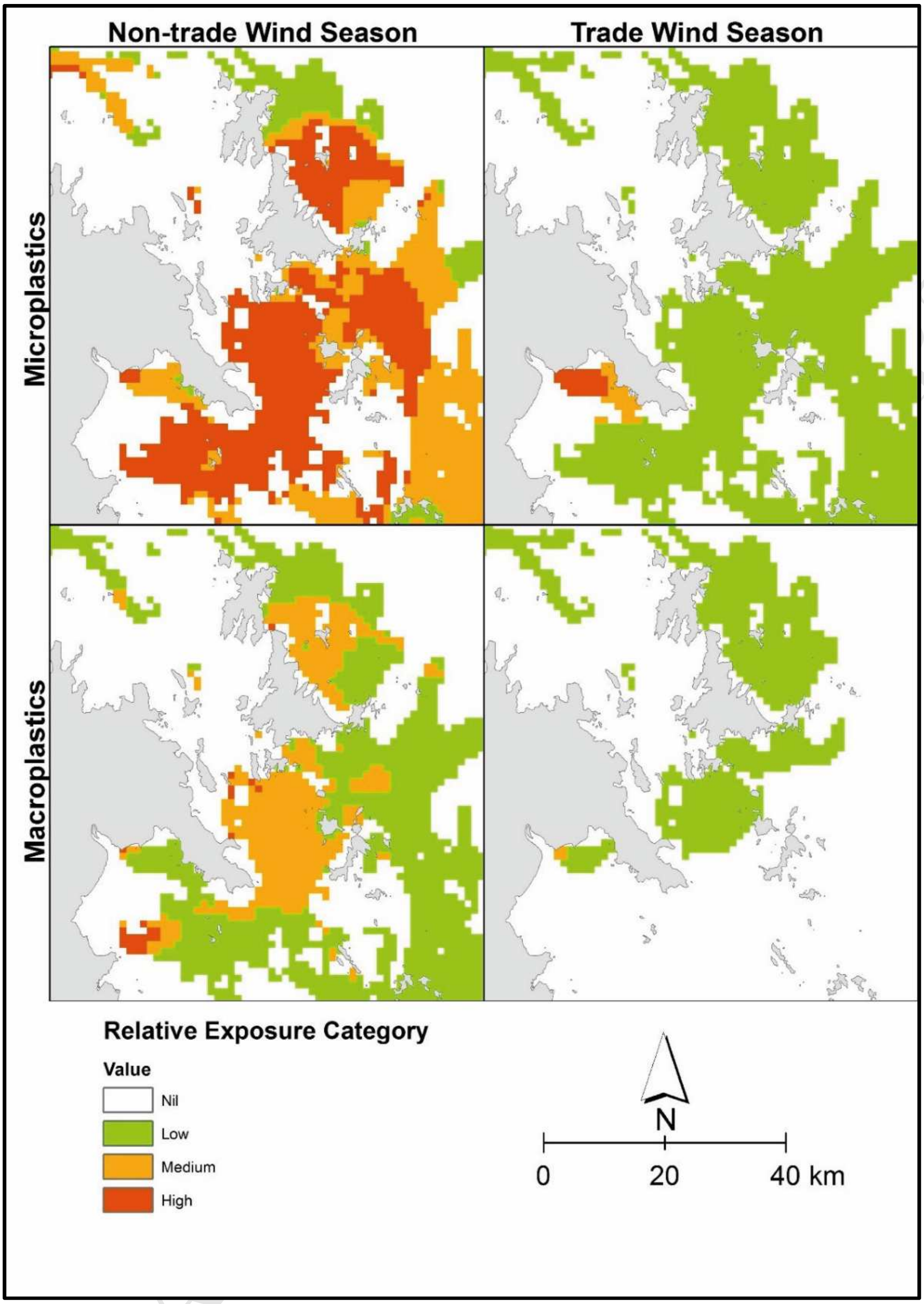
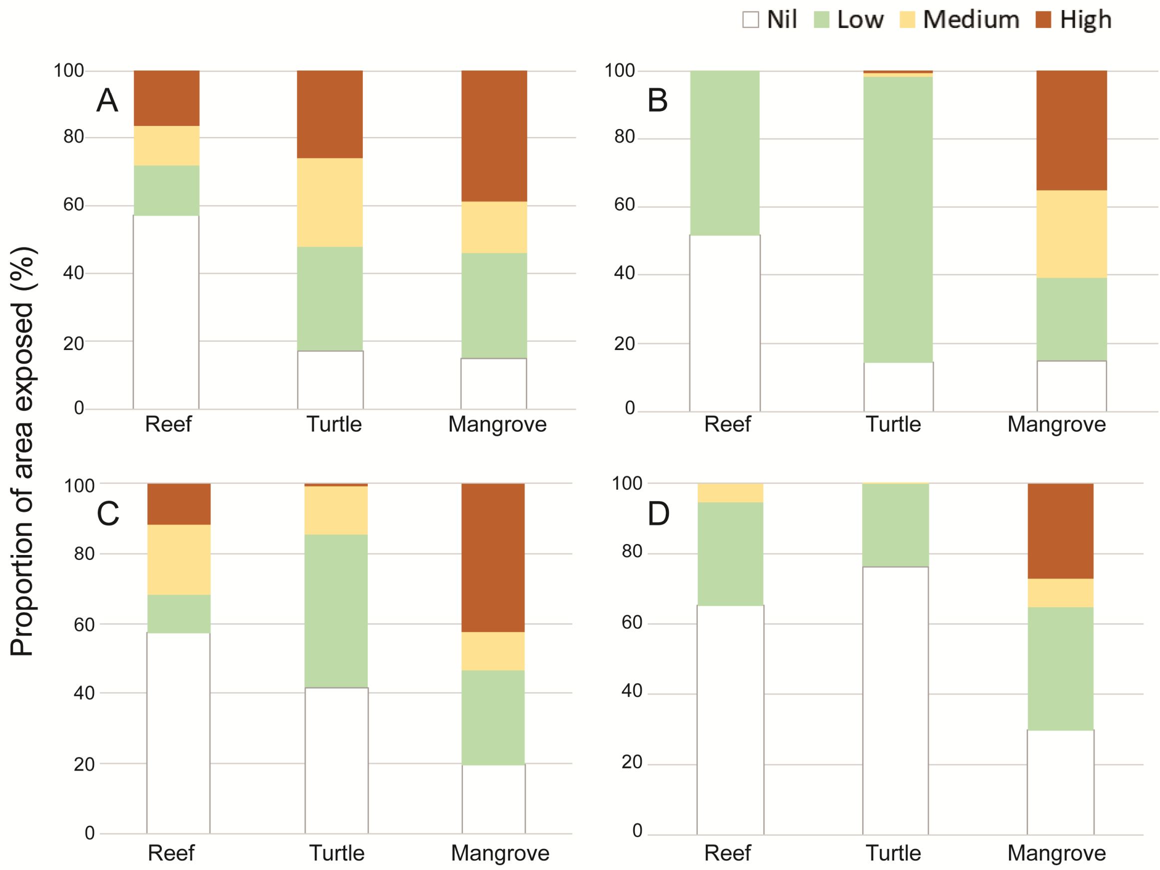
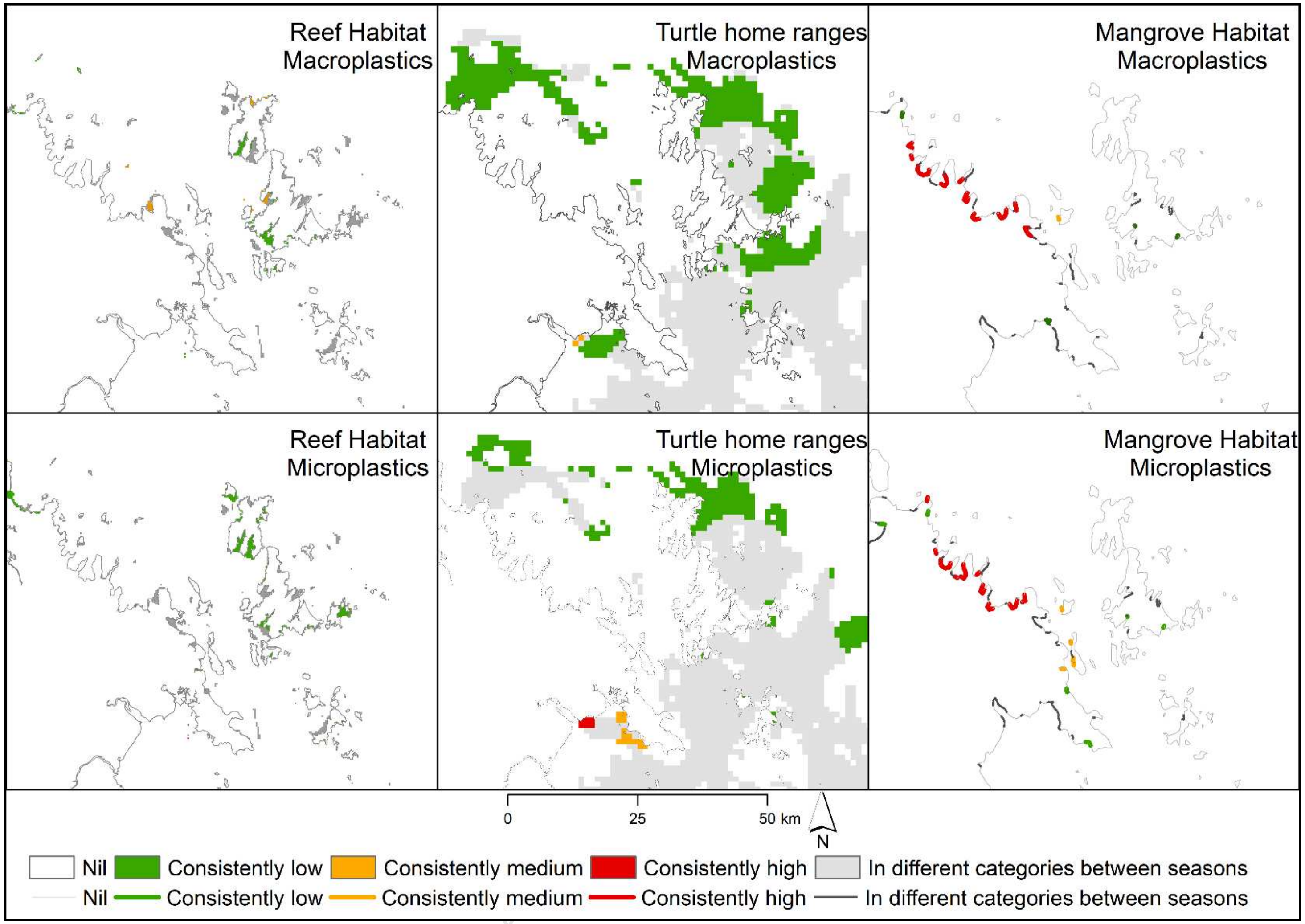


Figure 3: The relative exposure to macro- and microplastics for flatback turtle home ranges in the Whitsunday region. The equivalent map for coral reef and mangrove habitats can be found in the online appendix. Maps displaying the raw modelled values can also be found in the online appendix,



601

602 *Figure 4: Bar graph showing the proportion of habitat area in each threat category by season and plastic type. The threat*
603 *categories are colour coded as per the legend; nil in white, low in green, medium in yellow, high in red. A) Shows the areas*
604 *exposed to each category of microplastic exposure in the monsoon season, B) Shows the areas exposed to each category of*
605 *microplastic exposure in the trade wind season, C) Shows the areas exposed to each category of macroplastic exposure in*
606 *the monsoon season, D) Shows the areas exposed to each category of macroplastic exposure in the trade wind season.*

607


608 *Figure 5: Areas of coral reef habitat, mangrove habitat, and flatback turtle home range that are in a consistent exposure category across wind seasons.*

Table 1: Evidenced and speculated consequences of exposure to plastic pollution for coral reefs, marine turtles, and mangroves, for details of the data used please see online appendix 1. The consequences have been split into that of macroplastics (objects > 5 mm), and microplastics (objects < 5 mm).

Habitat or species	Consequence	Plastic type	Speculated or evidence of consequence	Exposure layers
Coral Reefs	Scouring/smothering of corals	Macroplastic	**Speculated** (Donohue et al. 2001; Goldberg 1997) **Evidence** (de Carvalho-Souza et al. 2018)	Settled macro
	Increased prevalence of disease	Macroplastic	**Evidence** (Lamb et al. 2018)	Settled macro
	Ingestion by animals in the habitat	Microplastic	**Evidence** (Gall and Thompson 2015; Hall et al. 2015)	Settled micro
	Invasive species	Macroplastic and microplastic	**Evidence** (Barnes 2002; Gregory 2009)	Settled macro and micro
Marine turtles	Gastrointestinal disruption after consumption	Macroplastic and microplastic	**Evidence** (Di Bello et al. 2013; Nelms et al. 2016; Parga 2012; Schuyler et al. 2014)	Suspended macro and micro
	Toxicological effects after consumption	Macroplastic and microplastic	**Speculated in turtles** **Evidence in other organisms** (Cole et al. 2015; Heindler et al. 2017; Rochman et al. 2013)	Suspended macro and micro
	Entanglement	Macroplastic	**Evidence** (Blasi and Mattei 2017; Duncan et al. 2017; Nelms et al. 2016; Wilcox et al. 2013)	Suspended macro
Mangroves	Scouring/smothering	Macroplastic	**Speculated** (Goldberg 1997; Smith 2012; Uneputty and Evans 1997)	Beached macro
	Changed community structure	Macroplastic	**Evidence** (Katsanevakis et al. 2007)	
	Ingestion by animals in the habitat	Microplastic	**Evidence** (Besseling et al. 2013; Gall and Thompson 2015; Van Cauwenberghe et al. 2015)	Beached micro
	Degradation of habitat quality	Microplastic	**Speculated** (do Sul et al. 2014)	
	Invasive species	Macroplastic and microplastic	**Evidence** (Barnes 2002; Gregory 2009)	Beached macro and micro

Table 2: The justification and importance of each source location in the plastic dispersal simulations (Figure 1).

Location	Rank of importance	Number of seeding points	Source justification
Airlie Beach	High	5	Large regional township (population 12,928; Census, 2016)
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	Tourist anchorage
North Hook Island	low	2	Tourist anchorage
Bowen	High	5	Regional township (population 9105; Census, 2016)
Outer Shipping lane	High	20 evenly spread every 10 km.	Objects lost or discarded from the 10s of thousands of ships that pass through annually (Maritime safety Queensland)

- Weather conditions strongly dictate plastic exposure patterns
- Variable winds in the monsoon season lead to higher exposure of plastic
- Strong south-easterly trade winds lead to plastic removal, therefore lower exposure
- Plastic types have different accumulation patterns at a small geographic scale

Declaration of interests

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

☐The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: