



Synthetic shorelines in New Zealand? Quantification and characterisation of microplastic pollution on Canterbury's coastlines

PJ Clunies-Ross, GPS Smith, KC Gordon & S Gaw

To cite this article: PJ Clunies-Ross, GPS Smith, KC Gordon & S Gaw (2016) Synthetic shorelines in New Zealand? Quantification and characterisation of microplastic pollution on Canterbury's coastlines, *New Zealand Journal of Marine and Freshwater Research*, 50:2, 317-325, DOI: [10.1080/00288330.2015.1132747](https://doi.org/10.1080/00288330.2015.1132747)

To link to this article: <https://doi.org/10.1080/00288330.2015.1132747>



Published online: 23 Mar 2016.



Submit your article to this journal [↗](#)



Article views: 2186



View related articles [↗](#)



View Crossmark data [↗](#)



Citing articles: 35 View citing articles [↗](#)

SHORT COMMUNICATION

Synthetic shorelines in New Zealand? Quantification and characterisation of microplastic pollution on Canterbury's coastlines

PJ Clunies-Ross^{a,b}, GPS Smith^c, KC Gordon^c and S Gaw^{a,b}

^aDepartment of Chemistry, University of Canterbury, Christchurch, New Zealand; ^bWaterways Centre for Freshwater Management, University of Canterbury, Christchurch, New Zealand; ^cDepartment of Chemistry, Dodd-Walls Centre, Otago University, Dunedin, New Zealand

ABSTRACT

Microplastics are persistent environmental contaminants found in marine environments worldwide. Microplastic particles isolated from coastlines in the Canterbury region of New Zealand were quantified and characterised. Sediment samples were collected from 10 locations representing exposed-beach, estuarine and harbour environments in both urban and non-urban settings. Particles were isolated from sediments using an NaCl density-separation procedure and quantified and characterised with a combination of optical/fluorescence imaging and micro-Raman spectroscopy. Microplastics were detected at eight out of 10 locations, at concentrations ranging from 0–45.4 particles kg⁻¹ of dry sediment. The majority of microplastics were identified as polystyrene (55%), polyethylene (21%) and polypropylene (11%). Microplastic concentrations in exposed-beach environments were significantly greater than in harbour and estuarine environments.

ARTICLE HISTORY

Received 25 July 2015

Accepted 21 November 2015

KEYWORDS

Canterbury; marine debris; microbead; microplastics; New Zealand; Raman

Introduction

Numerous studies over the past decade have shown that microplastics are distributed widely in the marine environment, accumulating in oceans, coastlines and freshwater environments around the world (Ivar do Sul & Costa 2014). Microplastics are defined by the US National Oceanic and Atmospheric Administration as plastic particles less than 5 mm in size (Arthur et al. 2009).

Microplastics may be preformed or derived from the degradation of larger macroplastic debris. Pre-manufactured microplastics or 'primary microplastics' include virgin plastic pellets, microbeads used in cosmetic products, and synthetic abrasives used in industrial cleaning applications (Fendall & Sewell 2009; Andrady 2011). Secondary microplastics are formed by physical, chemical or biological degradation of larger plastic debris (Cole et al. 2011). Degradation of plastic in the marine environment is primarily initiated by exposure to ultraviolet (UV) radiation from sunlight (Andrady 2011). The degradation process is exacerbated by physical abrasions with sediments and the hydrolytic properties of seawater (Moore 2008). Another common type of secondary microplastics comprises small

fibres derived from washing synthetic clothing that are not removed during wastewater treatment processes (Browne et al. 2011).

Microplastics can passively travel vast distances in the marine environment (Ryan et al. 2009). Their distribution is influenced by the plastic's ability to float, suspend or sink. This is largely determined by the density of the polymer. The fouling of plastics by microorganisms can cause particles to sink that would otherwise remain afloat (Andrady 2011). Once plastics have entered the marine environment, wind direction and currents subsequently influence their distribution. Onshore winds tend to force plastics back on to coastlines, whereas offshore winds push plastics into ocean current systems (Moore 2008). The resulting spatial and temporal variability in microplastic abundance is extremely difficult to predict.

Microplastics are available for uptake by a variety of marine animals across trophic levels (Browne et al. 2013; Rochman et al. 2013; Wright et al. 2013). Organisms that ingest microplastics may not obtain the required nutrition for activity and growth. Ingestion may also result in internal abrasions, false perceptions of satiation, occlusion of digestive processes and possible starvation (Moore 2008; Wright et al. 2013). Additionally, there are concerns that microplastics may act as a source of chemical contaminants. Plastics are typically produced from a range of toxic petroleum-based materials and additives (Lithner et al. 2011). As complete polymerisation is rarely achieved during the manufacturing process, unreacted monomers and chemical additives may leach from the material (Andrady 2011). Bisphenol A (BPA) and alkylphenol additives can have oestrogenic effects, while phthalate plasticisers have been associated with reduced testosterone production in exposed animals (Teuten et al. 2009).

Microplastics can also sorb and concentrate a range of organic contaminants from the water column including polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs) and organochlorine pesticides such as DDT (Teuten et al. 2009). There is growing evidence to suggest that particles may act as a vector to transfer sorbed contaminants to marine species upon ingestion (Wright et al. 2013). Fish exposed to environmentally relevant concentrations of marine microplastics have been found to bioaccumulate contaminants (Rochman et al. 2013). Humans consuming edible species may also be unwittingly ingesting these contaminants, as bivalves (Van Cauwenberghe et al. 2015) and fish (Lusher et al. 2013) have been found to contain microplastics in environmental settings.

To date, the majority of microplastics research has been conducted on coastlines in the Northern Hemisphere and within the great oceanic gyres (Ivar do Sul & Costa 2014). The number of studies conducted in Australasia and other regions in proximity to New Zealand is currently limited. Microplastic concentrations in Chilean and Australian coastlines have been reported to range from 11–30 particles 250 mL⁻¹ sediment with greater concentrations reported in densely populated locations (Browne et al. 2011). A subsequent volunteer survey conducted on the continental coastlines of Chile reported an average microplastic abundance of 27 particles m⁻² across 39 sites (Hidalgo-Ruz & Thiel 2013).

This is the first study to quantify and characterise primary and secondary microplastic pollution on New Zealand's coastlines. The Canterbury region provides an excellent study area to investigate microplastic pollution in New Zealand as it contains the nation's second largest city and has a diverse variety of coastal environments located in close proximity.

Materials and methods

Sample collection

Sediment samples were collected on 9 and 19 September and 1 October 2013 from 10 coastal locations in the greater Canterbury region representing exposed-beach, estuarine and harbour environments in both urban and non-urban settings (Figure 1). Samples were collected at each site by visually locating the most prominent strandline, characterised as the region with the greatest accumulation of washed-up debris. Three replicate samples were collected 50 m apart. A 0.25 m quadrat was placed at the centre of this zone and the first 2 cm layer of sediment was collected into glass jars. Large debris was removed with a 5.6 mm sieve. At Governors Bay, samples were collected at 20 m intervals due to the length of the beach. Samples were returned to the laboratory and dried at 70 °C. Non-synthetic clothing was worn during sample collection and analysis.

Isolation and identification of microplastics

A density separation method was adapted from techniques developed by Thompson et al. (2004), Claessens et al. (2011) and Vianello et al. (2013). For each sample, 500 mL of saturated NaCl solution was added to 200 g of homogenised sediment in a 1 L flask and manually shaken for 30 s before settling for at least 24 h before the supernatant was removed by suction. The separation process was then repeated with fresh NaCl solution, solutions combined and filtered through a 32 µm sieve, the lower size limit for microplastics in this study. Isolated particles were recovered on 47 mm glass microfibre filters (Whatman GF/C), transferred to Petri dishes and dried at 70 °C before microscopic analysis.

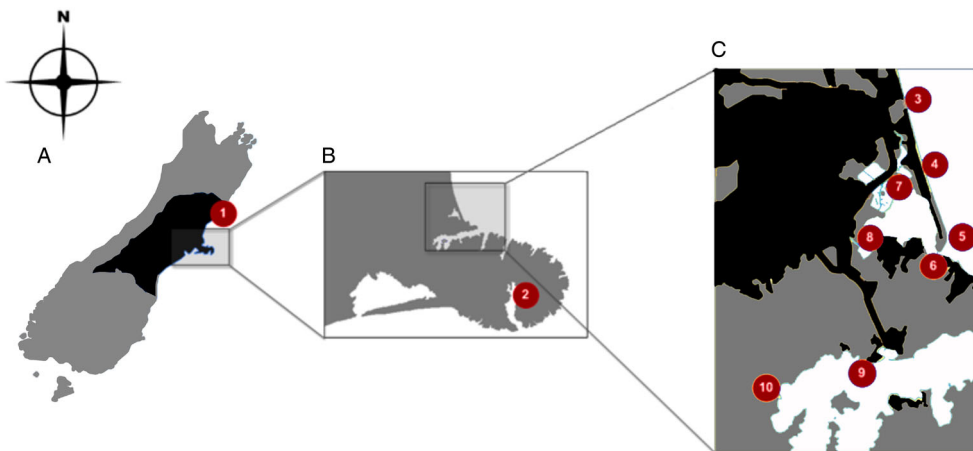


Figure 1. Sampling locations. **A**, Canterbury region shaded in black. Sample site 1 = Amberley beach; **B**, Banks Peninsula area. 2 = Akaroa; **C**, Christchurch City and Lyttelton Harbour. Urban areas coloured in black. Non-urban areas coloured in grey. Marine/estuarine environments coloured in white. 3 = New Brighton; 4 = South New Brighton; 5 = South Shore; 6 = Clifton; 7 = Estuary at Avon River mouth; 8 = Estuary at Heathcote River mouth; 9 = Corsair Bay; 10 = Governors Bay.

The efficiency of the density separation method was determined with the addition of reference plastic materials to sediment samples. A sample was spiked with 20 polyethylene microbeads (0.5 mm), five polyethylene pellets (4 mm) and five polycarbonate pellets (4 mm). Twenty-nine of 30 (97%) microplastic beads and pellets were recovered. Blank controls without sediment addition were used to measure possible contamination during experimental procedures. Small numbers of synthetic fibres were observed in a number of control and sediment samples indicating some contamination during the analytical procedure. This issue has been highlighted in other microplastic studies (Fries et al. 2013; Nuelle et al. 2014). Synthetic fibres were excluded from quantification for the purposes of this study. No contamination of other microplastics was detected.

Dried filter papers and associated isolated material were examined with the use of a Leica MZ10F fluorescence-coupled microscope. Filter papers were examined with 8–80 \times magnification in non-fluorescent mode to locate potential microplastics. The use of the UV filter (excitation filter 360/40 nm, barrier 420 nm) was also used to locate small potential microplastics containing fluorophores. Quantities of potential microplastics were counted and characterised as fragments, microbeads or pellets. Particles were also measured using Leica Application Suite software and grouped into the size categories: <1mm; 1–2 mm; or 2–5 mm size groups. The colour of particles was also noted.

The chemical composition of identified potential microplastics was determined with micro-Raman spectroscopy. The following parameters were used: laser wavelength $\lambda_0 = 785$ nm; 50 μm aperture; 10–20-fold magnification objective; 10–100 mW power; and spectra obtained in the range of 90–3200 cm^{-1} (Stokes shift). Reference spectra were obtained from verified samples of acrylonitrile butadiene styrene (ABS), low-density polyethylene (LDPE), high-density polyethylene (HDPE), polyethylene terephthalate (PET), polyvinyl chloride (PVC), polypropylene (PP), polystyrene (PS), polycarbonate (PC), polyurethane (PUR) and polyamide (PA). Particles of interest were located with the microscope and spectra obtained from surfaces unobscured by biofouling or organic debris. Molecular fingerprints of spectra captured were compared against reference polymers using Opus 6.5 software. A process of elimination was used in order to rule out possible polymers until a match was identified if possible. A number of suspected plastic particles could not be matched against reference spectra and were labelled as ‘suspected microplastics’ if the colour, shape and characteristic synthetic Raman peaks of particles were consistent with a plastic material. Microplastics were grouped into four categories: polyethylene (PE); polypropylene (PP); polystyrene (PS); and ‘suspected microplastics’.

Data analysis

The mean concentrations of particles by weight (particles kg^{-1} dry sediment) and area (particles m^{-2}) were calculated. Non-parametric statistical tests (Kruskal–Wallis and Mann–Whitney U) were used to compare microplastic concentration abundances in exposed-beach, harbour and estuarine environments.

Results

The density separation procedure used in this study was validated by the high recovery rate (97%) of microplastics from spiked sediment samples. A total of 127 suspected

microplastic particles were identified in samples using optical/fluorescence microscopy. Seventy-three of 127 (57.5%) isolated particles were confirmed to be microplastics with micro-Raman spectroscopy. Concentrations of confirmed microplastics ranged from 0–45.4 microplastics kg^{-1} dry sediment (Table 1). The highest concentration of 45.4 microplastics kg^{-1} dry sediment was measured at South Shore, an exposed-beach site in close proximity to Christchurch city.

The highest mean concentration of 21.2 ± 16.5 microplastics kg^{-1} was calculated for exposed-beach environments. Lower concentrations of 3.9 ± 3.5 and 1.7 ± 2.4 microplastics kg^{-1} were detected in harbour and estuarine environments, respectively. The concentrations of microplastic particles measured in exposed-beach, estuarine and harbour sediments were significantly different ($P = 0.014$). The mean concentrations measured in exposed-beach sediments were significantly different from harbour ($P = 0.019$) and estuarine ($P = 0.025$) environments.

The composition of the microplastics was identified by Raman spectroscopy. Examples of microplastics which share spectral features with polyethylene, polystyrene and polypropylene are shown in Figure 2. Of the 73 particles positively identified as microplastics, 54.8% were polystyrene. Polyethylene, polypropylene and suspected microplastics accounted for 20.5%, 11.0% and 13.7%, respectively. Many of the microplastics showed distinct signs of degradation and weathering. Fragments accounted for 86.3% of all microplastics. The remainder of the microplastics consisted mainly of pellets (11.0%). Two particles were classified as suspected microbeads (Figure 3). Fragments and microbeads were isolated only from exposed-beach sites.

Table 1. Microplastic concentrations in Canterbury's coastal environments. Results are reported as the mean microplastic concentration (\pm standard deviation), maximum concentration, site mean and microplastic concentration m^{-2} for each sample location.

Sample location	Site type	Substrate type	Mean microplastic concentration kg^{-1} dry sediment	Maximum concentration kg^{-1} dry sediment	Site mean	Microplastics m^{-2}
Amberley Beach	Exposed beach	Course sand + pebbles	3.3 ± 2.9	5.0	21.2 ± 16.5	177 ± 153
New Brighton		Medium sand	33.1 ± 11.5	40.3		1748 ± 609
South New Brighton		Medium sand	29.4 ± 13.2	40.1		1552 ± 695
South Shore		Fine sand	36.6 ± 14.3	45.4		1933 ± 757
Clifton Beach		Fine sand	3.3 ± 2.9	5.0		175 ± 151
Governors Bay	Harbour	Course sand + pebbles	Not detected	Not detected	3.9 ± 3.5	Not detected
Corsair Bay		Course sand + shells	5 ± 0.0	5.0		265 ± 3
Akaroa Harbour		Fine sand	6.7 ± 7.7	15.1		353 ± 407
Avon River mouth	Estuarine	Very fine sediment	Not detected	Not detected	1.7 ± 2.4	Not detected
Heathcote River mouth		Medium sediment + rocks	3.4 ± 2.9	5.1		178 ± 154

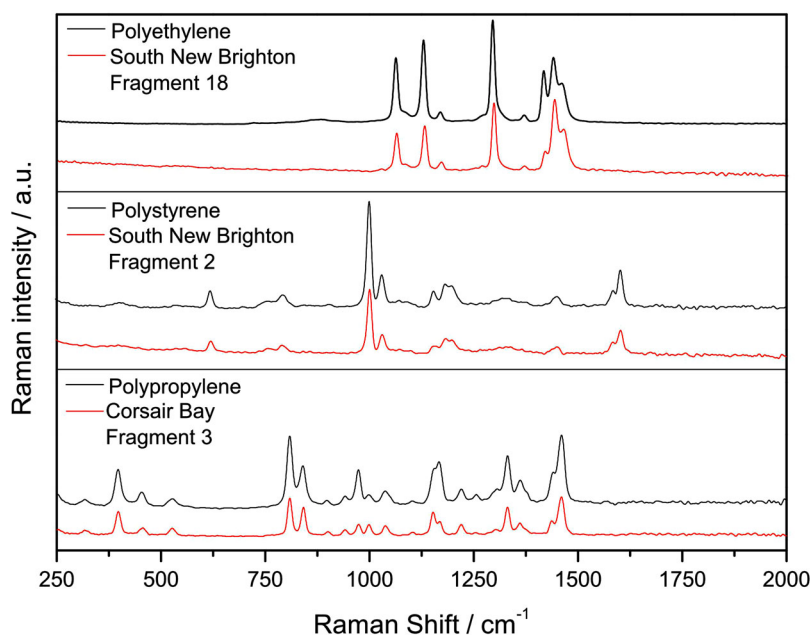


Figure 2. Raman spectra collected for **A**, polyethylene; **B**, polystyrene; and **C**, polypropylene. Reference material is shown in black. Raman spectra obtained from microplastics isolated from South New Brighton (A, B) and Corsair Bay (C).

The majority (58.9%) of microplastics were 2–5 mm in diameter; 24.7% were 1–2 mm and 16.4% were <1 mm in size. White microplastic accounted for 67.1% of confirmed microplastics. The remaining microplastics were clear (9.6%), blue (8.2%), red (5.5%), green (5.5%), yellow/orange (2.7%) and brown (1.4%).

Discussion

Microplastics were isolated from eight of the 10 coastal locations in Canterbury, New Zealand with concentrations in sediments ranging from 0–45.4 microplastics kg^{-1} . These concentrations are comparable to earlier international studies that have used a

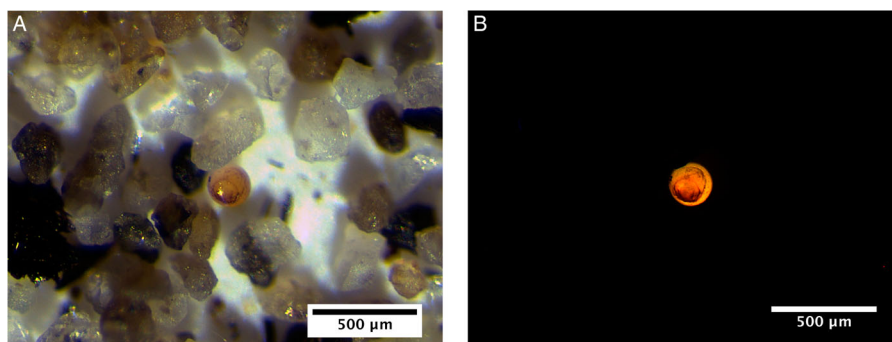


Figure 3. Suspected orange microbead and sand isolated from the exposed-beach coastline of New Brighton. Observed in **A**, optical; and **B**, UV fluorescence modes.

similar NaCl-based density separation technique. Thompson et al. (2004) reported concentrations of 8–86 microplastics kg^{-1} of dry sediment in the United Kingdom in the first published study. Subsequent studies reported concentrations of 0–12 microplastics kg^{-1} in Singapore (Ng & Obbard 2006) and 53–390 microplastics kg^{-1} in Belgium (Claessens et al. 2011). The highest concentrations reported to date for coastal sediments were measured in sediments collected from the Lagoon of Venice (672–2175 microplastics kg^{-1}) (Vianello et al. 2013).

Significantly higher microplastic concentrations were measured at exposed-beaches compared with harbour ($P = 0.019$) and estuarine ($P = 0.025$) sites. Higher concentrations were also typically measured at sites located in close proximity to the urban centre of Christchurch. Amberley Beach, located 40 km north of Christchurch city, was the only exposed-beach site located in a non-urban setting. Microplastic concentrations measured at this location were considerably lower than those observed at the urban sites of New Brighton, South New Brighton and South Shore. It is likely that the distance from the city and the substrate type are responsible for this observation. Low concentrations were also observed at Clifton Beach, an exposed-shore beach in close proximity to the city. This site had an extremely broad strandline and it is likely that microplastics are more widely distributed in this area resulting in the lower microplastic concentrations observed.

Spatial variability of concentrations observed between replicate samples was often high. This is not unexpected given that plastic distribution on coastlines is unlikely to be uniform. This pattern has been reported in the Belgian (Claessens et al. 2011) and Italian (Vianello et al. 2013) studies.

Considerably lower microplastic concentrations were detected in harbour sediments. Very little macroplastic debris was observed at harbour sites on the day of sampling and the low microplastic concentrations reflect this. The low concentrations measured in estuarine sites were unexpected. As large quantities of plastic waste were observed at both the Avon and Heathcote River mouths during field sampling, it was expected that microplastic concentrations at these locations would be high. However, the majority of plastic waste was found on the estuary bank and not within the observed strandline sampled. No particles were confirmed as microplastics at the Avon River mouth and low concentrations were measured at the Heathcote River mouth. It is conceivable that the sampling location chosen did not represent the appropriate area for plastic accumulation. Plastic debris may accumulate in zones higher than what was sampled during periods of large tidal movements or are flushed out to nearby coastal environments during tidal movements.

The majority of microplastics isolated were polystyrene, polyethylene and polypropylene. The greatest proportion (58.9%) of microplastics measured 2–5 mm in size. The lowest proportion (16.4%) measured <1 mm. The majority of particles isolated were white and clear. This is consistent with a review of earlier international studies reporting that the highest proportions of microplastics isolated from local coastal sediments are large, white and transparent (Hidalgo-Ruz et al. 2012). The size and colour of microplastic particles may contribute to their likelihood of being ingested by marine species. It has been suggested that fish may feed on white, brown and yellow microplastics that resemble prey (Shaw & Day 1994). The high abundance of these particles suggests that microplastics may be a hazard to marine species in the Canterbury region.

The majority of microplastics were fragments (86.3%), followed by pellets (11%). Two microbeads were isolated and may represent the first instance in which these particles have

been detected in local coastal sediments. The first bead was white and blue, approximately 250 µm in diameter and identified as polyethylene. The second was orange, highly fluorescent and the same size (Figure 3). The chemical composition of this bead was not conclusively identified due to high levels of noise in the Raman spectra generated.

The microplastic concentrations detected in this study are comparable to plastic pellet concentrations measured for the Canterbury coastline in 1978. Gregory (1978) reported concentrations ranging from 0 to >1000 pellets m⁻² at exposed-beach sites with the greatest concentrations measured at South New Brighton and South Shore. Pellets were infrequently detected in harbour environments. Consistent with the current study (Table 1), the greatest concentrations of plastic pellets were detected in exposed-beach sites located close to Christchurch.

Microplastic concentrations on the coastlines of Canterbury, New Zealand are likely to be strongly influenced by the urban population of Christchurch. The degraded nature of some particles suggests long residence times in the environment and the potential for widespread movement. Ocean currents surrounding New Zealand may also be a source of microplastic pollution for Canterbury's coastlines. Sampling conducted in the waters surrounding Australia reported average microplastic concentrations of 4256 particles km⁻² (Reisser et al. 2013).

Conclusion

This study has confirmed that primary and secondary microplastic pollution is present on New Zealand's coastlines at concentrations in the lower range of those measured internationally. The greatest proportion of microplastics comprised white polystyrene fragments 2–5 mm in size. Concentrations of microplastics measured from exposed-beach environments were significantly greater than those from estuarine and harbour environments. Additional studies are recommended to further investigate the presence and impacts of microplastic pollution in New Zealand.

Acknowledgements

Associate Editor: Associate Professor Conrad Pilditch.

Disclosure statement

No potential conflict of interest was reported by the authors.

References

- Andrady AL. 2011. Microplastics in the marine environment. *Mar Pollut Bull.* 62:1596–1605.
- Arthur C, Baker J, Bamford H. 2009. Proceedings of the International Research Workshop on the Occurrence, Effects, and Fate of Microplastic Marine Debris; 2008 Sept 9–11; University of Washington Tacoma, Tacoma, WA. NOAA Technical Memorandum NOS-OR&R-30. Silver Spring, MD: NOAA Marine Debris Division.
- Browne MA, Crump P, Niven SJ, Teuten E, Tonkin A, Galloway T, Thompson R. 2011. Accumulation of microplastic on shorelines worldwide: sources and sinks. *Environ Sci Technol.* 45:9175–9179.

- Browne MA, Niven SJ, Galloway TS, Rowland SJ, Thompson RC. 2013. Microplastic moves pollutants and additives to worms, reducing functions linked to health and biodiversity. *Curr Biol.* 23:2388–2392.
- Claessens M, De Meester S, Van Landuyt L, De Clerck K, Janssen CR. 2011. Occurrence and distribution of microplastics in marine sediments along the Belgian coast. *Mar Pollut Bull.* 62:2199–2204.
- Cole M, Lindeque P, Halsband C, Galloway TS. 2011. Microplastics as contaminants in the marine environment: a review. *Mar Pollut Bull.* 62:2588–2597.
- Fendall LS, Sewell MA. 2009. Contributing to marine pollution by washing your face: microplastics in facial cleansers. *Mar Pollut Bull.* 58:1225–1228.
- Fries E, Dekiff JH, Willmeyer J, Nuelle M-T, Ebert M, Remy D. 2013. Identification of polymer types and additives in marine microplastic particles using pyrolysis-GC/MS and scanning electron microscopy. *Environ Sci Processes Impact.* 15:1949–1956.
- Gregory MR. 1978. Accumulation and distribution of virgin plastic granules on New Zealand beaches. *New Zeal J Mar Freshwater Res.* 12:399–414.
- Hidalgo-Ruz V, Gutow L, Thompson RC, Thiel M. 2012. Microplastics in the marine environment: a review of the methods used for identification and quantification. *Environ Sci Technol.* 46:3060–3075.
- Hidalgo-Ruz V, Thiel M. 2013. Distribution and abundance of small plastic debris on beaches in the SE Pacific (Chile): a study supported by a citizen science project. *Mar Environ Res.* 87–88:12–18.
- Ivar do Sul JA, Costa MF. 2014. The present and future of microplastic pollution in the marine environment. *Environ Pollut.* 185:352–364.
- Lithner D, Larsson Å, Dave G. 2011. Environmental and health hazard ranking and assessment of plastic polymers based on chemical composition. *Sci Total Environ.* 409:3309–3324.
- Lusher AL, McHugh M, Thompson RC. 2013. Occurrence of microplastics in the gastrointestinal tract of pelagic and demersal fish from the English Channel. *Mar Pollut Bull.* 67:94–99.
- Moore CJ. 2008. Synthetic polymers in the marine environment: a rapidly increasing, long-term threat. *Environ Res.* 108:131–139.
- Ng KL, Obbard JP. 2006. Prevalence of microplastics in Singapore's coastal marine environment. *Mar Pollut Bull.* 52:761–767.
- Nuelle M-T, Dekiff JH, Remy D, Fries E. 2014. A new analytical approach for monitoring microplastics in marine sediments. *Environ Pollut.* 184:161–169.
- Reisser J, Shaw J, Wilcox C, Hardesty BD, Proietti M, Thums M, Pattiaratchi C. 2013. Marine plastic pollution in waters around Australia: characteristics, concentrations, and pathways. *PLoS One.* 8: e80466.
- Rochman CM, Hoh E, Kurobe T, Teh SJ. 2013. Ingested plastic transfers hazardous chemicals to fish and induces hepatic stress. *Sci Rep.* 3: article 3263.
- Ryan PG, Moore CJ, van Franeker JA, Moloney CL. 2009. Monitoring the abundance of plastic debris in the marine environment. *Philos T R Soc B.* 364:1999–2012.
- Shaw DG, Day RH. 1994. Colour- and form-dependent loss of plastic micro-debris from the North Pacific Ocean. *Mar Pollut Bull.* 28:39–43.
- Teuten EL, Saquing JM, Knappe DRU, Barlaz MA, Jonsson S, Bjorn A, Rowland SJ, Thompson RC, Galloway TS, Yamashita R, et al. 2009. Transport and release of chemicals from plastics to the environment and to wildlife. *Philos T R Soc B.* 364:2027–2045.
- Thompson RC, Olsen Y, Mitchell RP, Davis A, Rowland SJ, John AWG, McGonigle D, Russell AE. 2004. Lost at sea: where is all the plastic? *Science.* 304:838–838.
- Van Cauwenberghe L, Claessens M, Vandegehuchte MB, Janssen CR. 2015. Microplastics are taken up by mussels (*Mytilus edulis*) and lugworms (*Arenicola marina*) living in natural habitats. *Environ Pollut.* 199:10–17.
- Vianello A, Boldrin A, Guerriero P, Moschino V, Rella R, Sturaro A, Da Ros L. 2013. Microplastic particles in sediments of Lagoon of Venice, Italy: First observations on occurrence, spatial patterns and identification. *Estuar Coast Shelf S.* 130:54–61.
- Wright SL, Thompson RC, Galloway TS. 2013. The physical impacts of microplastics on marine organisms: a review. *Environ Pollut.* 178:483–492.