# MARINE DEBRIS POLLUTION IN THE PACIFIC

## Literature review

Laysan Island marine debris (photo by: David Liittschwager)

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# LITERATURE REVIEW

## Marine debris pollution in the Pacific

> *'..plastic items are important to society; however, there is something fundamentally different between the problem of marine debris and other current societal dilemmas. Unlike turning on an electric light or taking an airplane journey, the emission (in this case of debris to the oceans) is not directly linked to the benefit.'* – Richard Thompson in Koelmans *et al.* (2014)

## 1. Background

### 1.1. Human pressure on marine environment

Rapidly expanding human populations and associated economic growth and overconsumption is resulting in serious degradation of the natural environment human survival depends on (Vitousek *et al.*, 1997; Sanderson *et al.*, 2002; Orr, 2004; Alroy, 2010; Branch *et al.*, 2013). Almost half of the global human population currently lives within 150km of the coast (UN Atlas of the Oceans, 2014). This results in severe pressures being placed on marine and coastal environments. Anthropogenic impacts on oceans include physical alterations of the coasts and seafloor, as well as chemical and biological contamination through sewage, industrial wastes and agricultural discharges, dredging, desalination, shipping, and fossil fuel and ore extraction. These pressures, together with overfishing, by-catch, destructive fishing methods (e.g. blast fishing), introduction of invasive species, boat strikes, acoustic pollution, climate-related changes (i.e. ocean acidification, sea level rise, freshwater inundations, cyclones) can cause structural changes in marine communities and the loss of genetic variability and other side-effects of human interference with exceptionally complex ocean ecosystems (Gray *et al.*, 1979; Goldberg, 1995; Vitousek *et al.* 1997; Islam and Tanaka, 2004; Pauly *et al.*, 2005; Panigada *et al.*, 2006; Crain *et al.*, 2008; Halpern *et al.*, 2008; Ramirez-Llodra *et al.*, 2011).

Various types of marine pollution are of particular environmental concern. Recently, contamination of oceans by solid wastes has been recognized as a serious threat to marine life, and consequently for humanity as well (UNEP, 2009).

### 1.2. Marine debris pollution - definition

Any persistent, manufactured or processed solid material that enters the ocean environment from any source is considered marine debris or marine litter (Coe and Rogers, 1997). It includes all items discarded, disposed of, or abandoned in the marine and coastal environment (UNEP, 2005). Marine debris is also generated by natural events (including tsunami, floods and cyclones), when various man-made objects are washed from the coastal areas into the marine environment (Lebreton and Borrero, 2013). Once at sea, these objects can interfere with marine biota causing a range of impacts (Derraik, 2002).

Humankind has used oceans for disposing of its waste for more than two millennia, believing that oceans have the capability of self-cleansing. This behavior did not have major environmental impacts while the coastal populations were small and waste was mostly biodegradable (Gregory and Andrady, 2003; Sheavly and Register, 2007). However, exponentially growing human population, and the post-World War II economic and consumption 'boom' are considered to be directly linked with the escalation of marine debris pollution problems. This set of conditions lead to increased waste generation, some of which is non-biodegradable or degrade very slowly (i.e. plastic), so when it reaches the ocean it tends to accumulate (Katsanevakis, 2008).

The problem of marine debris pollution was first recognised in 1970s' academic publications as a potentially serious environmental problem. The scientific community reported the occurrence of man-made objects in different marine environments; floating on the surface (Heyerdahl, 1971; Carpenter *et al.*, 1972; Venrick *et al.*, 1973; Colton *et al.*, 1974), on the sea floor (Holmstrom, 1975; Feder *et al.*, 1978) and as shore accumulations (Scott, 1972, Cundell, 1973; Gregory, 1977). Several authors reported the incidence of ingestion of plastic particles by fish (Carpenter *et al.*, 1972; Kartar *et al.*, 1973) and seabirds (Rothstein, 1973; Hays and Cormons, 1974).

Despite the evidence of plastic accumulation in the marine environment and biota, marine debris was considered only an eyesore by the plastics industry of that time, comprising only a small proportion of all litter and causing no harm to the environment (Ferguson, 1974, in Derraik, 2002, p. 842). Waldichuk (1978), then a leading Canadian marine scientist, in his report on global marine pollution stated that 'while this type of pollutant is largely cosmetic in character and affects mainly the amenities, there are other uses of the sea which are also affected' (p. 31), listing the impacts of plastic on boating in the form of propeller entanglement. However, marine debris pollution is now currently acknowledged as a severe environmental hazard and raises a great deal of concern due to its cumulative persistent nature and associated toxicity, and known, as well as its yet unknown, impacts on marine ecosystems and humans (Derraik, 2002; Gregory and Andrady, 2003; Thompson *et al.*, 2004; Sheavly and Register, 2007; Moore, 2008; Barnes *et al.*, 2009; Gregory, 2009; Ryan *et al.*, 2009; Thompson *et al.*, 2009a; Kershaw *et al.*, 2011; Koelmans *et al.*, 2014).

## 1.3. Sources of marine debris

Discarded or lost anthropogenic objects enter the oceans through a variety of human activities, such as street and beach littering, improper waste management, fishing and shipping, offshore drilling, at sea accidents, and natural disasters (UNEP, 2005). These sources of marine debris can be broadly divided into 2 major groups: land-based and ocean-based sources (Ribic *et al.*, 1992; Rees and Pond, 1995; Frost and Cullen, 1997; Cunningham and Wilson, 2003). Gregory (1999b) proposed three possible sources of the marine debris found deposited onshore in the SW Pacific. These are: (i) local and land-based, (ii) nearby offshore maritime activities (e.g. fisheries), (iii) oceanic or pelagic debris drifting from afar, regardless of the original source. The latter division is more precise, but the simple one is more commonly and widely used. Additionally, Williams *et al.* (2003) noted that pelagic litter, in its origin, is still one of the two, either land-based or ocean-based.

Interestingly, marine environments and their health, productivity and biodiversity are considerably more affected by human activities on the coasts and further inland than by marine-based activities (UNEP, 1995). In the early endeavours to combat marine debris pollution, most attention was given to the activities occurring at sea. However, in the early 1990s it was estimated that as much as 80% of marine debris originated from land and only 20% from marine-based sources (Gregory and Andrady, 2003; Trouwborst, 2011). Generation of marine debris is a consequence of poor waste management (both on land and ships), illegal human activities (littering and dumping) and incidental pollution (natural events, accidents and unintentional loss). Most of it can be attributed to the consumption of food, drinks and smoking-related products and the use of oceans for transport and harvesting food items (Sheavly and Register, 2007).

### *1.3.1. Land-based activities*

Land-based debris mostly originates from densely populated coastal urban areas, including metropolitan and industrial zones, and is sourced from the diverse human activities occurring within these zones (Gregory and Andrady, 2003). Generated litter is transported to the marine environment by winds and rains. The most important sources of marine debris include the following categories (Gregory, 1999b; Sheavly and Register, 2007; NOAA, 2008, UNEP, 2009):

- **Coastal areas** including beaches, streets, piers, harbours, marinas, docks and riverbanks:

Marine debris from these sources usually is usually derived from public littering and inappropriate solid waste management services. The rubbish is left behind or dumped in public areas, such as streets, parks, parking lots and beaches (Sheavly and Register, 2007), especially if there were no waste receptacles (Bator *et al.*, 2011). This litter is then blown by the wind or washed away by rains and snowmelt into nearby waterways and sewage systems or directly into the ocean. Waterways can carry litter sourced from distant inland areas as well (Gregory and Andrady, 2003; Sheavly and Register; 2007, Moore, 2008). Leite *et al.* (2014) found significant relationship between beach litter density and proximity of the beach to the urban centre Salvador in Brazil.

Several authors examined the input of litter via rivers (Armitage, 2007; Moore *et al.*, 2011; Carson *et al.*, 2013a; Morritt *et al.*, 2014). Carson *et al.* (2013a) used two debris-retention booms to measure the quantity of surface debris entering Hilo Bay in Hawaii. They estimated that booms retained debris from 10.2% of land area and was generated by approximately 4400 residents. In 205 days, booms retained almost 30kg of man-made objects, majority of which was plastic.

Moore *et al.* (2011) employed different devices and nets to sample the litter entering Los Angeles Basin through two major rivers. The samples were collected through the whole river profile, from the bottom to the surface and from the middle to the edge of the channel. They found that as much as 30.4 metric tons of plastic objects and particles entered Los Angeles basin in 72 hours.

Armitage (2007) used litter traps to measure litter input in Cape Town, South Africa. He found that litter generation depended greatly on socio-economic conditions and available waste management services. Low income areas of Cape Town produced more waste than high income areas, largely because removal of domestic waste was more efficient in richer areas. However, poorer areas tend to be more densely populated as well.

- **Discharges of untreated municipal sewage and storm water**:

Humans have been using drainage systems and waterways to dispose of various wastes since ancient times. Evidence from Mesopotamia and Indian village Rakhigarhi shows that sewage systems were used more than 4000 years ago (Lofrano and Brown, 2010). As a consequence, waterways are the main conduit of pollutants from land to oceans, from municipal to industrial and agricultural wastes (Islam and Tanaka, 2004).

Large quantities of litter reach the oceans through discharges of untreated municipal sewage and storm water (Pruter, 1987; Nollkaemper, 1994). Some old sewage systems are combined systems where sanitary sewage and stormwater runoff are combined within the same infrastructure. During heavy rains, to avoid overloading, overflow mechanisms of these systems are set to bypass the treatment plant and discharge the entire content (including sanitary wastes and street litter) directly into rivers and oceans. This type of waste presents a substantial component of marine debris (Nollkaemper, 1994; Sheavly and Register, 2007; Katsanevakis, 2008; NOAA, 2008). Williams and Simmons (1999) found that riverine litter from the River Taff in South Wales was mainly derived from overflows of combined sewage systems and discharge from illegal dump sites. Sewage-related litter, such as sanitary towels, comprised 23% of all riverine litter.

- **Municipal landfills** and **illegal dump sites** located on the coast or inland that are connected to a waterway:

Landfills or illegal dump sites situated on the coasts or near rivers are another important source of marine debris. If not properly managed through burial, rains and winds can cause runoffs of lightweight waste to waterways and the marine environments. This also applies to carelessly handled waste during collection, transport and processing in a treatment facility. Waste from illegal dumping to water bodies (rivers, lakes) or on the coast, has even more potential to become marine debris because it is not under any form of control (Pruter, 1987; Sheavly and Register, 2007; Katsanevakis, 2008; Barnes *et al.*, 2009).

- **Industrial waste**:

Commonly reported industrial waste in the marine environment comes from the plastics industry in the form of plastic pellets, which are raw material for producing plastic goods (Hays and Cormons, 1974). They enter marine environments from all sectors of the plastics industry, from manufacturing sites, transporters and packagers to processors, usually during loading and unloading of trucks, trains and ships and through the outfalls of plastics factories (Fig. 1.3.1.) (Redford *et al.*, 1997).

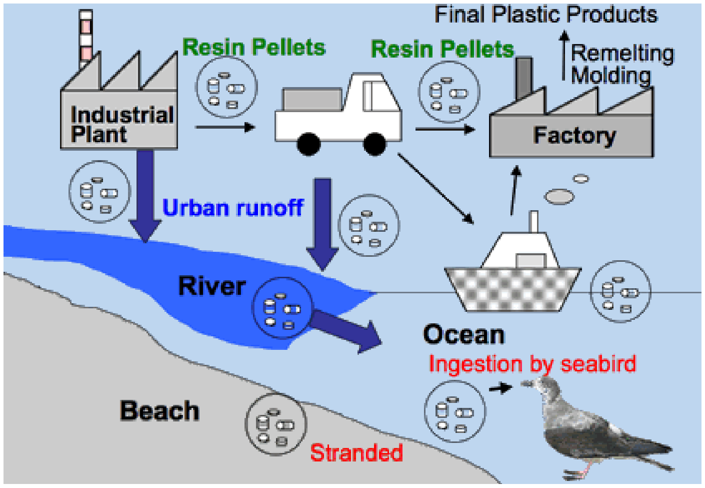

Fig. 1.3.1. Fate of plastic pellets in the marine environment (adopted from International Pellet Watch (2014)).

Ship-breaking industry contributes to marine debris pollution as well (UNEP, 2009). Alang-Sosyia yard in Western India is one of the world's largest ship breaking yards. Considerable amounts of scraping waste from ship demolition accumulate on the shore and consequently enter the ocean. Reddy *et al.* (2003) surveyed the shore and collected ship demolition debris, such as rubber, insulation material, paper, metal, glass, plastics, textiles, paints and thermocol, in following densities: 15.63 kg $m^{-2}$ in Sosyia and 10.19 kg $m^{-2}$ in Alang (Reddy *et al.*, 2003). In their study on microplastic pollution in the same area they found that on average 1kg of sediment contained 81.43mg of anthropogenic particles (Reddy *et al.*, 2006).

- **Medical waste**:

Clarification of how medical waste enters the ocean environment was not found in the available literature dealing with marine debris pollution. Potentially, it happens as a result of random disposal or uncontrolled dumping and consequent runoff. The majority of medical waste is similar in composition to domestic waste, but 10-25% of it is hazardous, and includes sharps (syringes) and infectious waste, and poses health risks (ICRC, 2011).

Philipp *et al.* (1994) reported a fourfold increase in the occurrence of medical waste on the coastline of England and Wales between 1991 and 1993. In three annual surveys of 15% of the British coastline by Norwich Union Coastwatch, volunteers found 185.2, 196.1 and 769.2 medical items per 1000km in 1991, 1992 and 1993, respectively. In 1993 they collected as many as 94 syringes.

- **Coastal tourism** involving recreational visitors and beach-goers:

Clean beaches are important for tourism (Ryan and Swanepoel, 1996); however tourism and the recreational use of coasts have been recognised as an important source of marine debris (Sheavly and Register, 2007; UNEP, 2009). Tourists, beachgoers and recreational shore fishermen tend to litter beaches and picnic and recreation areas. Several authors investigated the relationship between the amount of litter found on beaches utilized by coastal tourism (Martinez-Ribes *et al.*, 2007; Ariza *et al.*, 2008; Tsagbey *et al.*, 2009; Portz *et al.*, 2011). All authors found that the quantity of beach debris increases during the tourist season with greater beach use.

### *1.3.2. Ocean-based activities*

Marine pollution caused by solid waste discharged from ships is regulated through the MARPOL 73/78 convention, Annex V, however persistent anthropogenic materials regularly reach marine environments. Human activities at sea frequently result in accidental or intentional loss of gear, waste and other man-made items. Topping *et al.* (1997) noted that 76% of 739 recorded fishing trips (1990-1991) resulted in ships waste being dumped into the ocean. Along with operational waste from ocean-based activities, general waste from ships also enters the ocean. It should be noted that most reports on ocean-based debris, apart from fishing-related waste and debris, are fairly outdated (Horsman, 1982; Pruter, 1987, Pearce *et al.*, 1992; Topping *et al.*, 1997). Relevant marine-based sources are as follows (Gregory and Andrady, 2003; Sheavly and Register, 2007; NOAA, 2008; UNEP, 2009):

- **Shipping** (merchant, public transport, pleasure, naval and research vessels):

Merchant shipping, as reported in the late 1980s, and based on reports from 1975-1985, contributed to marine debris pollution more than other ship-related maritime activities. A vast majority of debris created by commercial shipping (98%) was associated with cargo and operational waste, and included dunnage, shoring, pallets, wires and plastic covers. Only 2% of waste was crew-related (plastics) (Pruter, 1987).

Merchant ships may lose cargo at sea during heavy weather. The World Shipping Council (2011) reported annual loss of approximately 675 containers, lost due to various factors including catastrophic events. Both containers and their content present a safety or pollution hazard at sea. For example, the Chinese oil giant Sinopec lost containers, carrying plastic pellets, as the aftermath of a severe Typhoon Vincente, which hit southern China in July 2012. The accidental spillage of 150 tonnes of pellets seriously compromised the state of local fish farms and marine environment in general (FEHD, 2012). More recently, Danish shipping company Maersk lost 520 containers in a single event caused by rough weather in February 2014 (Seatrade Global, 2014).

Military dumping at sea used to be a common practice. Keller *et al.* (2010) identified military waste on the bottom off Cape Flattery, Washington, as the main contributor to metal waste contamination of the seafloor. Military items included ammunition boxes, helmets, rocket boosters and launchers. Some areas in the Mediterranean, such as Adriatic Sea and central Mediterranean, are particularly contaminated by ammunition, most of which presents a serious environmental hazard and concern for public safety. Many of dumped bombs, grenades, torpedoes and ship mines still hold explosive charge (Valkovic *et al.,* 2010).

- **Fishing and aquaculture activities**:

In contrast to other ocean-based sources of marine debris pollution, fishing-related debris has been extensively studied due to its regularly reported impacts on marine wildlife (Gregory, 2009; Laist, 1997). Prior to the 1940s, fishing gear was made of natural plant (cotton, linen, hemp, manila) or animal fibres (silk, hair). With the introduction of more durable synthetic materials in the late 1940s fishing has become much more efficient. The shift from natural to synthetic fibres was completed by the 1970s for most fishing nations (Pruter, 1987; von Brandt, 2005).

Abandoned, lost or otherwise discarded fishing gear (i.e. fishing nets, ropes, fishing lines, trawl floats, traps and pots, crates and boxes) now comprise a considerable proportion of marine debris due to the exceptional durability of synthetic materials (Gregory and Andrady, 2003; Sheavly and Register 2007; Katsanevakis, 2008). Fishing-related marine debris originates from both commercial and recreational fishing activities. Walker *et al.* (1997) monitored marine debris on the Bird Island in the Southern Ocean from 1990 to 1995 and found that 76% of all debris was long liner fishing line. Donohue *et al.* (2001)

determined trawl netting as the most frequent type of debris (88%) damaging the coral reefs in Hawaii, followed by monofilament gillnet (34%) and maritime line (23%). Kim *et al.* (2014) estimated that fisheries in South Korea discard 11,436 tonnes of traps and 38,535 tonnes of gill-nets in the coastal waters annually. In contrast, Watters *et al.* (2010) attributed marine debris, collected in the 1990s and 2007 off central California, almost entirely to recreational fishing, as monofilament fishing line from rod-and-reel fishing was the predominant debris item.

Additionally, aquaculture produces marine debris as well. In some areas in Japan where oyster cultivation is common, plastic pollution of the marine environment can be quite extensive. Fujieda and Sasaki (2005) collected 245,656 stranded debris items from 48.6km of coastline and 99.5% of it consisted of polystyrene fragments. Hinojosa and Thiel (2009) surveyed surface waters off southern Chile and found that ~80% of debris they collected was polystyrene most likely derived from local mussel farming activities.

- **Offshore mining and extraction** (oil and gas platforms and associated supplying vessels):

The offshore petroleum industry contributes to marine debris pollution as an ocean-based source as well. Operational waste and equipment enter the ocean due to improper disposal practices, per operational practices or rough weather (NOAA, 2008). The consequences of severe accidents on offshore platforms, such as the Deepwater Horizon explosion and oil spill in 2010 in the Gulf of Mexico, not only affect marine environment through oil pollution, but also created marine debris (Carmichael *et al.*, 2012).

### *1.3.3. Natural events*

Natural storm-related hazards, such as tsunami, typhoon and floods, associated with strong winds and waves, generate large amounts of marine debris. King tides, which flood low-lying islands, have similar impacts (Bruce Kijiner, RMI, pers. comm.). Man-made objects in coastal areas, which are exposed to storms or flood events, may be released into the ocean (UNEP, 2009). The Boxing Day Tsunami in 2004, apart from having numerous human victims, caused much damage to coral reefs in Indonesia, Thailand, the Andaman Islands and Nicobar Islands, Sri Lanka and the Maldives, smothering and destroying corals by debris and dead coral rubble (Ramachandran *et al.*, 2005; Wilkinson *et al.*, 2006).

The Tohoku tsunami, which hit Japan in March 2011, created a large amount of marine debris as well. Over half million houses and buildings were damaged and destroyed and almost 25 million tonnes of debris were washed into the ocean, including cars, boats and other objects (Oh, 2011). Lebreton and Borrero (2013) estimated that the influx of debris into the North Pacific in this single event is likely to equal thousands of years of 'normal' debris input.

## 1.4. Composition of marine debris

### *1.4.1. By usage*

Composition of marine debris indicates potential sources of pollution. Human activities in a certain area may reflect and help drive the differences in the local marine debris composition (Kershaw *et al.*, 2011). The prevailing types of debris are derived from human consumption patterns (e.g. food, drink, cigarette products and associated packaging), ocean transport and fishing activities. Sheavly and Register (2007) stated that littering is the core of the marine debris problem, in particular from food and beverage packaging.

Categorization of marine debris is not universally standardised and debris identification is a fairly subjective process, thus litter categories are variable from study to study (Williams *et al.*, 2003). Land-sourced and ocean-sourced debris items are sometimes clearly distinguishable, but in some cases it is not possible to unambiguously ascribe an item solely to one source (Gregory and Andrady, 2003; Williams *et al.*, 2003). For example, buoys and fishing nets are undoubtedly ocean-based debris, but items related to everyday human activities, such as consumption or hygiene, may originate from both land and oceans. Galley wastes from ships are similar in composition to land-based domestic refuse, which makes it difficult to identify the origin of those items once they become marine debris (Gregory and Andrady, 2003). Conversely, some wastes, specifically related to ships, may also come from land-based sources, such as ship scraping waste from ship recycling yards (oiled sponges, insulation material, paint, etc.) (Reddy *et al.*, 2003).

Sourcing marine debris, albeit being difficult, complex and often imprecise, is one of the most important tasks of marine debris management. Reliable attributions to the origin of debris are crucial. There are three main factors to consider when ascribing a source to marine debris: *identification*, *function* and *quantity* (Williams *et al.*, 2003). Identifying the item is critical, but easily fallible. The actual function of a debris item is not necessarily equal to its primarily intended function, i.e. it can have a secondary purpose. For example, containers cut in half are most likely used as bailers in boats, oil funnels or paint holders. Sometimes local knowledge plays an important role in debris identification (Williams *et al.*, 2003). For example, 65 socks and 214 balloons, collected in 23 surveys from a beach in Coffs Harbour (NSW, Australia), are not likely to be related to fishing; however local fisheries scientists implied that socks are frequently used for bait and balloons for fishing for a local fish (luderick: pers. comm. Matt Broadhurst and Paul Bucher; Markic and Smith, unpub.). Furthermore, the quantity of a certain item is also an important factor when relating it to a possible source. A single or rare finding is not considered sufficient to indicate a significant source (Williams *et al.*, 2003).

It is generally accepted that more populated coastal areas produce more land-based debris, while typical ocean-based debris is more commonly found on remote shores (Barnes *et al.*, 2009; Hammer *et al.*, 2012). Land-based debris mostly includes domestic and industrial waste, such as plastic bottles, packaging, fast food containers, plastic bags and plastic sheeting (Gregory and Andrady, 2003). Typical ocean-based debris usually comprises lost fishing gear and operational waste from fishing and shipping activities (Vauk and Schrey, 1987). One of the indicators of *ex situ* debris, brought by ocean currents and waves from offshore source, is the abundance of pumice on the shoreline (Ryan and Moloney, 1990).

Land-based debris related to consumption, sewage, hygiene, household, recreation, tourism and religious events was commonly documented as the prevailing type of beach debris in populated areas worldwide (e.g. Ross *et al.*, 1991; Thornton and Jackson, 1998; Zhou *et al.*, 2011; Jayasiri *et al.*, 2013; Kordella *et al.*, 2013; Gago *et al.*, 2014). Tsagbey *et al.* (2009) found a strong proportional relationship between the increase of tourist pressure and the amount of beach litter present in Ghana and, accordingly, most litter was related to tourist consumption (packaging and disposables). Martinez-Ribes *et al.* (2007) found that in the high tourist season in the Balearic Islands, a popular tourist destination in the Mediterranean, cigarette butts were the most common debris item, and they accounted for up to 46% of all litter. Similarly large proportion of cigarette butts in beach debris was obtained in Rio Grande do Sul (39%) (Portz *et al.*, 2011) and Armacao dos Buzios (~43%) (Oigman-Pszczol and Creed, 2007) in Brazil, and Obhur in Saudi Arabia (52%) (Kitto *et al.*, 2011). Foster-Smith *et al.* (2007) reported as much as 70% of all debris in Cable Beach in Broome, Western Australia, to be composed of cigarette butts. Cahoon (1990) collected 1564 aluminium cans from a dredge spoil island in Masonboro Sound in Southeast North Carolina from 2.5km of shoreline in 6.5 hours. He related the cans to recreational boaters, since 87% of the cans were alcoholic beverage cans.

Fishing-related debris, as previously mentioned, is typically the principal component of beach debris in remote isolated areas, such as polar and subpolar regions. It includes long-line fishing line, netting, ropes, buoys and various floats, fish crates, plastic straps, light-sticks and other items related to commercial fishing, deliberately or inadvertently released into the ocean. Substantial amounts of fishing debris were recorded on the beaches of Amchitka island in the North Pacific (Merrell, 1984), Chatham and Stewart Islands off New Zealand (Gregory, 1999a), Inaccessible Island in the south Atlantic (Ryan and Watkins, 1988) and several locations in the Southern Ocean, including Macquarie and Heard Island (Slip and Burton, 1991), Bird Island in South Georgia archipelago (Walker *et al.*, 1997), Livingston Island in South Shetlands archipelago (Torres *et al.*, 1997) and Falkland Islands (Otley and Ingham, 2003). This debris is related to local or nearby fisheries (Walker *et al.*, 1997; Gregory 1999a).

However, Benton's findings (1995) are not entirely in accordance with the generality that ocean-based debris is more common on remote shores and land-based near population centres. He compared the quantity and composition of debris found on the beaches of Pitcairn Islands, remote atolls in the South Pacific, and Inch Strand beach in populated and industrialized Ireland. The quantities of debris were similar, but the composition was not entirely as expected. Plastic bags, food wraps and disposable nappies were absent on the atolls, while in Ireland they were present in the following percentage: 12, 4 and 1, respectively. Buoys were more abundant on the atolls (8 and 19%), than in Ireland (3%). However, glass bottles made up to 17-18% of litter on the atolls, while only 6% in Ireland. Contrarily, rope accounted for 12% of all debris on Inch Strand and only 5-6% on the atolls. Therefore, the atolls were not contaminated mainly by typical ocean borne debris, such as nets and ropes, but it was more diverse. Nevertheless, most debris presumably originated from ships.

Fishing debris can be deposited in situ on beaches as well, dumped or left behind by fishermen. Taffs and Cullen (2005) reported increased quantities of fishing items on beaches in NSW, Australia, used for fishing and boat launching. In that case, fishing debris is of land origin, which can confound analysis of this type of samples.

An interesting study on submerged marine debris in the river Thames demonstrated that large amounts of litter enter the marine environment unnoticeably, along the river bottom (Morritt *et al.*, 2014). The authors set fyke nets on the bottom of the river and in less than 3 months captured 8490 various anthropogenic items. Most common items were general plastics, food wrappers and containers, sanitary items and tobacco packaging.

Studies of the ocean and sea floor indicate that both ocean- and land-related debris accumulate there. Commonly reported bottom debris is derelict fishing gear, such as lobster traps (Chiappone *et al.*, 2002; Uhrin *et al.*, 2014) and crab pots (Bilkovic *et al.*, 2014), fishing nets (Donohue *et al.*, 2001; Dameron *et al.*, 2007) and fishing lines (Hess *et al.*, 1999; Watters *et al.*, 2010; Pham *et al.*, 2013). However, many authors reported other types of debris as well. Galgani *et al.* (1995a) hauled mainly plastic bags (95% of all bottom debris) from the Bay of Biscay, France. Nagelkerken *et al.* (2001) attributed 74% of litter found on the bottom of Curacao coastal area to food consumption. Richards and Beger (2011) surveyed the bottom of the Majuro lagoon, Marshall Islands, and determined 78.7% of macro-debris to be household-related. They recorded items such as bricks, office chairs, nappies, rope, prams, shopping trolleys, rubber mats, car tyres, car bodies, paint tins, machinery, roofing iron and a 44 gallon drum. Fishing-related debris accounted for only 1% of all debris in this tropical atoll lagoon.

Surface debris often consists of small plastic fragments and pellets (Doyle *et al.*, 2011; Goldstein *et al.*, 2013a; Reisser *et al.*, 2013). Reisser *et al.* (2013) determined the possible origin of plastic particles by identifying the polymer type, which then indicated the potential identity and function. They attributed majority of sampled plastic particles, mainly made of polyethylene and polypropylene, to degradation of single-use disposable packaging and fishing equipment. Styrofoam was also present and the authors related those particles to packaging and fishing as well.

Other floating debris, larger in size and detectable from a distance, comprises both land-based and ocean-based items. In Straits of Malacca and Bay of Bengal, Ryan (2013) visually identified more than 18,000 items along 3275km of combined transect length and determined 92.3% of debris as packaging, the majority of which was polystyrene, bags and food wraps. Conversely, Pichel *et al.* (2007) reported that almost 74% of all large floating debris, observed in aerial surveys in 2005 in the northeast Pacific north of Hawaii, were fishing floats.

### *1.4.2. By material*

Marine debris consists of any man-made material, including plastic, rubber, glass, metal, textile, processed wood and construction materials. By far the most dominant component of marine debris (60-80%) are persistent synthetic materials most of which are composed of plastics (Derraik, 2002; Gregory and Andrady, 2003). Plastics can be both positively and negatively buoyant, and dominate marine debris in all ocean environments. According to Andrady (2011), 62% of globally produced plastics are positively buoyant.

Plastic materials were reported as the main constituent of beach debris in the vast majority of scientific publications on a global scale, from densely populated to remote and isolated areas, including the very early ones (Dixon and Cooke, 1977; Vauk and Schrey, 1987; Willoughby, 1986; Gabrielides *et al.*, 1991; Wade *et al.*, 1991; Golik and Gertner, 1992; Slater, 1992; Garrity and Levings, 1993; Frost and Cullen, 1997; Haynes, 1997; Madzena and Lasiak, 1997; Walker *et al.*, 1997; Bowman *et al.*, 1998; Kusui and Noda, 2003; Abu-Hilal and Al-Najjar, 2004; Claereboudt, 2004; Ivar do Sul and Costa, 2007; Morishige *et al.*, 2007; Al-Shwafi and Ahmed, 2011; Smith, 2012; Topçu *et al.*, 2013; Eriksson *et al.*, 2013). In some coastal areas it was documented that plastics comprised more than 90% of all marine debris, namely Hawaii (Cooper and Corcoran, 2010, Ribic *et al.*, 2012a), California (Moore *et al.*, 2001), Honduras (Cruz *et al.*, 1990), South Africa (Ryan and Moloney, 1990), Sable Island in the north Atlantic (Lucas, 1992) and Livingston Island in the south Atlantic (Torres *et al.*, 1997); Belgium coast (van Cauwenberghe *et al.*, 2013), and Coffs Harbour in Australia (Smith and Markic, 2013). Cooper and Corcoran (2010) studied accumulation rates of plastic debris on Maha'ulepu beach in Hawaiian island of Kauai. They sampled beach debris daily for 11 day and collected 6,082 plastic pieces from a quadrat 1x5m and 3cm of depth. The estimated accumulation rate they provided was 484 pieces day$^{-1}$, which means that almost 100 plastic fragments strands daily on 1m$^2$ of that beach.

Floating debris is often almost entirely composed of plastics (Goldstein *et al.*, 2013a; Eriksen *et al.*, 2013; Ryan, 2014). In many studies of marine debris on the ocean floor, plastics were the prevalent material as well, in both shallow areas (Nagelkerken *et al.*, 2001; Donohue *et al.*, 2001; Katsanevakis and Katsarou, 2004) and deep sea floor, canyons and ocean trenches (Galgani *et al.*, 2000; Miyake *et al.*, 2011; Bergman and Klages, 2012; Guven *et al.*, 2013; Ramirez-Llodra *et al.*, 2013). However, bottom debris tends to be more diverse regarding the composition by material than floating debris. For example, Katsanevakis and Katsarou (2004) found 25.7% of marine debris in Greek shallow coastal sea was composed of metal items (Al-cans, anchors, chains, pots, etc). Nagelkerken *et al.* (2001) recorded 20% of glass and 17% of metal in shallow subtidal off beaches in Curacao. Al-Najjar and Al-Shiyab (2011) found more metal items (41% - Al cans) contaminating the sea floor of Gulf of Aqaba (Jordan, Red Sea), than plastic (38%) and glass (17%). Richards and Beger (2011) recorded 46% metal and 13% glass debris. Thirty-one percent of metal items were aluminium cans and the rest were various metal objects, from food containers to roofing iron. Metal materials on the sea bottom are often abundant it fishing areas (Katsanevakis and Katsarou, 2004; Watters *et al.*, 2010; Bilkovic *et al.*, 2014). Morishige *et al.* (2007) studied accumulation of beach debris on an isolated Hawaiian island (Tern Island) for 16 years. The majority of debris were plastic items (71% plastic and 6% Styrofoam), but they also recorded 17% of debris as composed of glass.

Rubber materials in the ocean and coasts are usually reported in small quantities (Morishige *et al.*, 2007; Smith 2012; Ribic *et al.*, 2012a; Martin, 2013, Schlining *et al.*, 2013). They often include vehicle tyres (Katsanevakis and Katsarou 2004; Keller *et al.*, 2010; Richards and Beger, 2011) and rubber balloons (Lucas ,1992; Martin, 2013; Liu *et al.*, 2013; O’Shea *et al.*, 2014).

Paper is also typically a minor component of marine debris, however sometimes greater records are documented. Kordella *et al.* (2013) found 13-18% of paper on Greek beaches and Khairunnisa *et al.* (2012) reported 17% of paper on Malaysian beaches, where it consisted mostly of wet newspapers which were used as mats for picnickers. Kordella *et al.* (2013) stated that the possible reason behind such large paper content on Greek beaches is in fact a volunteers’ misidentification. This was also the case with cigarette butts from Brazilian beaches, which were incorrectly placed in the paper category (e.g. Oigman-Pszczol *et al.*, 2007). Cigarette filters, although they are covered by paper, are mainly made of cellulose acetate, which is a type of synthetic polymer (Andrady, 2003).

Styrofoam (i.e. extruded polystyrene foam) is a regularly reported debris item and it is often placed in a category separate from general plastics (Santos *et al.*, 2009; Khairunnisa *et al.*, 2012; Debrot *et al.*, 2013a), sometimes due to its prevalence (Smith 2010). It is often difficult to determine the source of styrofoam due to its fragmented state (Rosevelt *et al.*, 2013), but, when identifiable, it can be linked to fishing floats (Khordaghui and Abu-Hilal, 1994; Fujieda and Sasaki, 2005) and food containers (Garrity and Levings, 1993).

Most authors focus on plastic debris as the most hazardous and prevalent material of all marine debris components. Plastic debris can be divided in macro- and micro-plastic. Microplastic can further be divided into primary and secondary microplastics according to their origin (Koelmans *et al.*, 2014). Plastic fragments and microplastic (<5mm) have raised much concern in the recent years. They comprise much of debris in all ocean compartments, including beaches (Martins and Sobral, 2011; Hidalgo-Ruz and Thiel, 2013), ocean surface (Eriksen *et al.*, 2013) and subsurface (Desforges *et al.*, 2014), and ocean bottom (van Cauwenberghe *et al.*, 2013; Vianello *et al.*, 2013). The plastics industry contributes to the pollution from plastic pellets. They enter the ocean through outfalls from plastic manufacturing plants, via rivers, and from loss during loading, transport and offloading (trucks, trains, ships). They were also used as ball bearings for moving cargo (Pruter, 1987).

## 1.5. Plastics

### *1.5.1. Plastics: definition, production, types and applications*

Plastics are defined as "inexpensive, lightweight, strong, durable, corrosion-resistant materials, with high thermal and electrical insulation properties" (Thompson *et al.*, 2009a, p. 2154). The term *plastic* originates from Greek word *plastikos* and means 'pertaining to molding, fit for molding' (Online Etymology Dictionary, 2014). It refers to the properties of 'a wide range of materials that at some stage in manufacture are capable of flow such that they can be extruded, moulded, cast, spun or applied as a coating' (Thompson *et al.*, 2009b, p. 1973). There are two types of plastics: **thermoplastics**, which can be repeatedly formed into various shapes using heat and pressure (and are easily recyclable), and **thermosets** which, after solidification, cannot be re-melted or reshaped (e.g. epoxy and polyurethane) (Andrady, 2003).

Plastics are made by polymerisation of monomers obtained from crude oil and gas. Depending on the conversion routes from fossil fuel to useful monomers (Fig. 1.5.1), plastics can be divided into polyolefins and aromatic polymers, which are made from a lighter or heavier component of the fossil fuel, respectively (Andrady, 2003).

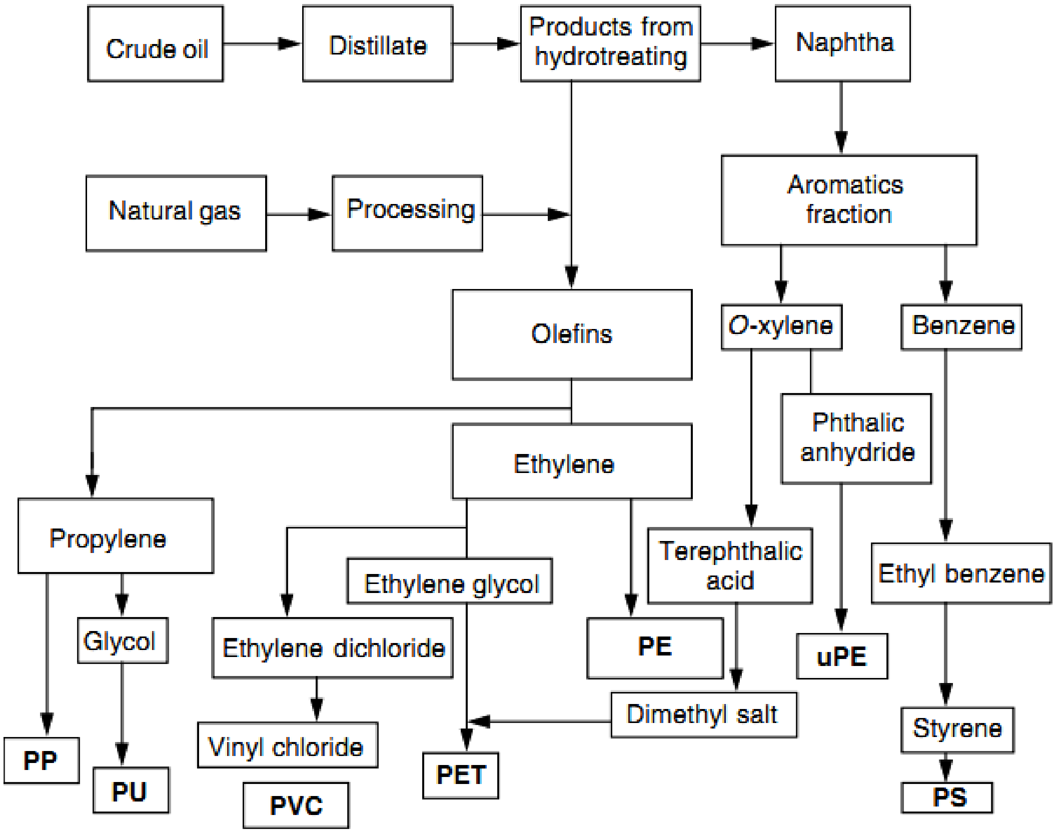

Fig 1.5.1. Conversion routes of the common plastics (Adopted from Andrady, 2003).

Raw plastic materials (virgin plastic polymers) are susceptible to deterioration in the environment, thus the addition of various chemicals (stabilisers) is required to improve their performance regarding resistance and durability (Andrady, 2003; Thompson *et al.*, 2009a). These stabilisers serve to prevent processes such as the formation of biofilm on polymer surface (biocides), photodegradation (UV stabilizers) and autoxidation (antioxidants) (Andrady, 2003). Other additives improve or change the properties of a plastic material according to specific needs, and these include pigments, reinforcing fillers, plasticizers and flame retardants (Table 1.5.1).

Table. 1.5.1. Common plastic additives and their functions (adopted from Andrady, 2003).

| Additive type | Function | Example |
|---|---|---|
| Antiblocking agent | Prevents sticking of thin plastic sheets to each other, or "blocking" | Quartz or silica in polyethylene |
| Antioxidans | Reduces the rate of autoxidation of the plastic at service temperature | Metal deactivators, peroxide decomposers |
| Antistatic agent | Prevents charges on polymer surface leading to static discharge | Quaternary ammonium salts in rigid PVC |
| Biocide | Prevents growth of microorganisms on plastics | Phenols and chlorinated phenols in coatings |
| Blowing agent | Used to create polymeric foams | Inert gases and AIBN*that decompose into $N_2$ on heating |
| Inert filler | Reduces the cost of formulation and changes the color | Chalk used in plastic formulations |
| Reinforcing filler | Increases the modulus and other properties of a polymer | Carbon black in rubber formulations |
| Coupling agent | Promotes better adhesion between phases in filled and glass-fiber-reinforced plastics | Organosilanes, titanates, and zirconates |
| Curing agent | Crosslinks the polymer | Sulfur or organic sulfur compounds in rubber |
| Flame retardant | Reduces the flammability of plastics products | Borates, and organophosphorous compounds |
| UV stabilizer | Minimizes the solar UV-B induced degradation of plastics outdoors | Hindered amines and light absorbers |
| Impact modifier | Increases the impact resistance of plastics | Rubber and thermoplastics in epoxy resin |
| Lubricant | Minimizes internal and external friction leading to degradation | Ethylene(bis)stearamide used in rigid PVC |
| Pigments | Colors plastic product | Inorganic pigments, carbon black, and organic pigments |
| Plasticiser | Softens the plastic and makes it more processable | Phthalates in rigid PVC compounds |

* AIBN – 2,2' azobisisobutyronitrile

Plastics materials are highly diverse (Andrady, 2003). The most widely used types of plastic are polypropylene (PP) and polyethylene (PE) (high density (HDPE), low density (LDPE) and linear low-density polyethylene (LLDPE)). Other types include poly(vinyl chloride) (PVC), poly(ethylene terephthalate) (PET), polystyrene (PS and PS-E (PS-expanded)) and polyurethane (PUR) (Fig. 1.5.2.). Their specific gravities range from 0.83 to 1.37 (Note: different materials are positively, neutrally or negatively buoyant with respect to the water with the specific gravity 1) (Table 1.5.2.). Furthermore, almost 40% of all plastics are manufactured for packaging (Fig. 1.5.3.) (Plastics Europe, 2013).

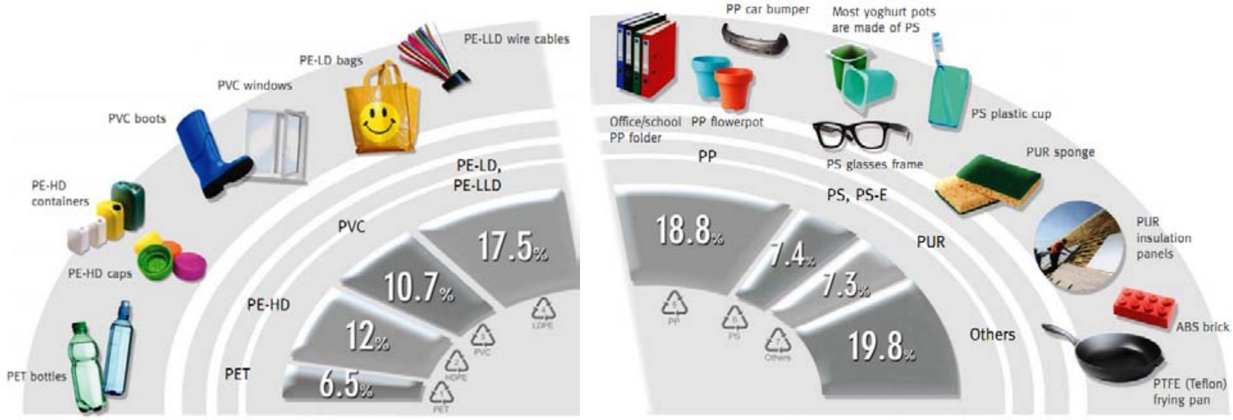

Fig. 1.5.2.. European plastics demand by resin type in 2012 (Plastics Europe, 2013).

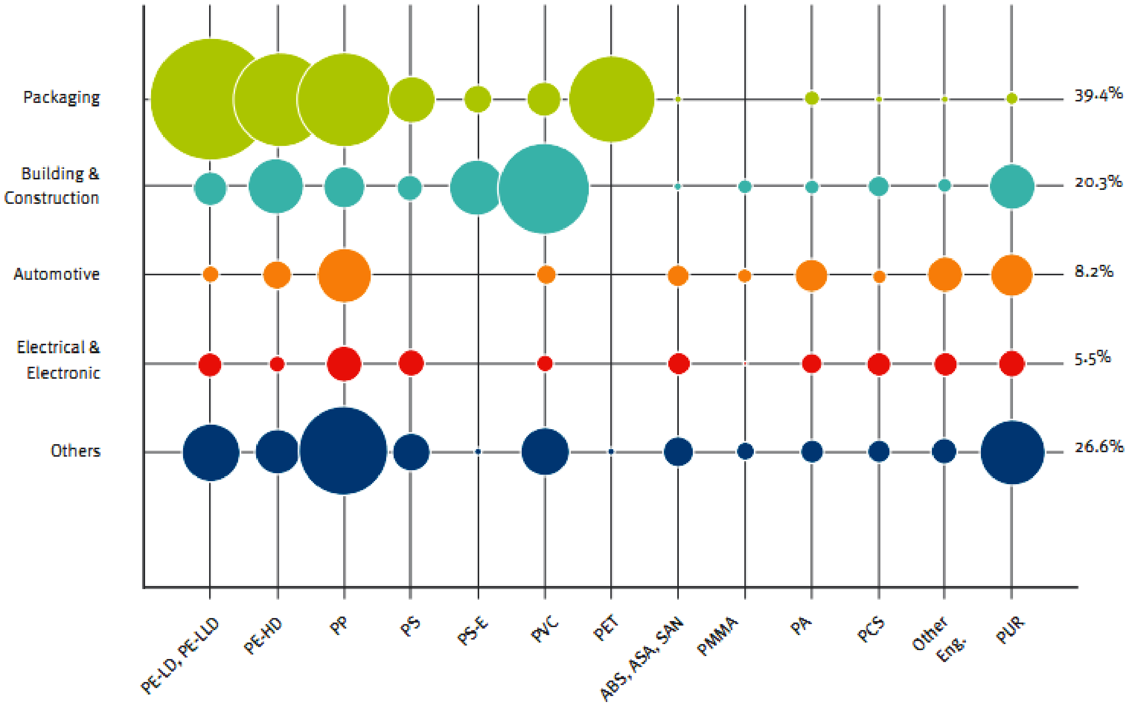

Fig. 1.5.3. European plastics demand by market in 2012 (segment and resin type) (Plastics Europe, 2012)

Table 1.5.2. Specific gravity of common plastics (after Andrady, 2011).

| Plastic class | | Specific gravity |
|---|---|---|
| Polypropylene | PP | 0.83-0.85 |
| Low-density polyethylene | LDPE, LLDPE | 0.91-0.93 |
| High-density polyethylene | HDPE | 0.94 |
| Polystyrene | PS | 1.05 |
| Thermoplastic polyester | PET | 1.37 |
| Poly(vinyl chloride) | PVC | 1.38 |

Once discarded in the environment, plastic materials undergo deterioration due to environmental factors such as ‘solar radiation, heat/cold, moisture (solid, liquid, and vapor), oxygen, and atmospheric contaminants’ (Andrady, 2003, p. 313). Apart from photodegradation and oxidation, degradation of plastics also occurs due to mechanical abrasion. It is worth noting that degradation of plastics, except for expanded polystyrene, is slower in water than in dry land environments, due to lower temperatures and decreased exposure to UV radiation in the water (Gregory and Andrady, 2003). Furthermore, biofouling protects the plastics from photodegradation as well (Andrady, 1990). Even though synthetic materials are subjected to mechanical degradation and biodegradation in the marine environment, the process of biodegradation of plastics lasts considerably longer than that of natural materials (Table 1.5.3.).

Table 1.5.3. Marine debris biodegradation time line (Mote Marine Laboratory, 1993)
(Retrieved from CMORE, 20 July, 2014)

| Item | Time to degrade |
|---|---|
| Paper towel | 2-4 weeks |
| Newspaper | 6 weeks |
| Apple core | 2 months |
| Cardboard box | 2 months |
| Waxed milk carton | 3 months |
| Cotton gloves | 1-5 months |
| Photo-degradable beverage holder | 6 months |
| Wool gloves | 1 year |
| Plywood | 1-3 years |
| Painted wooden sticks | 13 years |
| Plastic bags | 10-20 years |
| Tin can | 50 years |
| Disposable diapers | 50-100 years |
| Plastic bottle | 100 years |
| Aluminium can | 200 years |
| Plastic beverage holder | 400 years |
| Monofilament fishing line | 600 years |
| Glass bottle and jars | undetermined |

### *1.5.2.. History of plastics*

The first fully artificial plastic polymer, thermoset plastic named Bakelite, was synthesized by Leo Baekeland in his garage in New York in 1907 (Andrady, 2003). An example of Bakelite product was the old black dial phone. Among the first thermoplastics was vinyl plastic derived from ethylene, made after the 1930s (Andrady, 2003). PVC was known at the time, but did not have useful application except for waterproofing of fabrics. The common rigid PVC, used mainly in construction, was developed after the World War II. Polyethylene was produced in 1933 at Imperial Chemical Industries (ICI) research laboratories, but it was patented only in 1936. Polystyrene was also discovered in the 1930s and was produced in Germany and the United States, while Styrofoam (extruded polystyrene foam) was invented in 1954 (Hammer *et al.*, 2012). Nylon was introduced in 1939 at the World’s Fair in New York and used mainly in tire reinforcement, parachute fabric, toothbrushes and women’s stocking (Andrady, 2003). Polypropylene was invented in 1954 by Giulio Natta, which is now, together with polyethylene, the most common plastic material. Both are known as commodity plastics (Andrady and Neal, 2009).

Mass production of plastics started after the Second World War (Carpenter and Smith, 1972; Andrady, 2003). Since then, the global production has increased from 1.7 Mtonnes in 1950s to 288 Mtonnes in 2012 (Fig. 1.5.4.a), demonstrating positive correlation with the global human population growth from 2.5 to current 7.1 billion, respectively (World Bank, 2014). China and Europe were the major plastic producers in 2012 (Fig. 1.5.4.b) (PlasticsEurope, 2013).

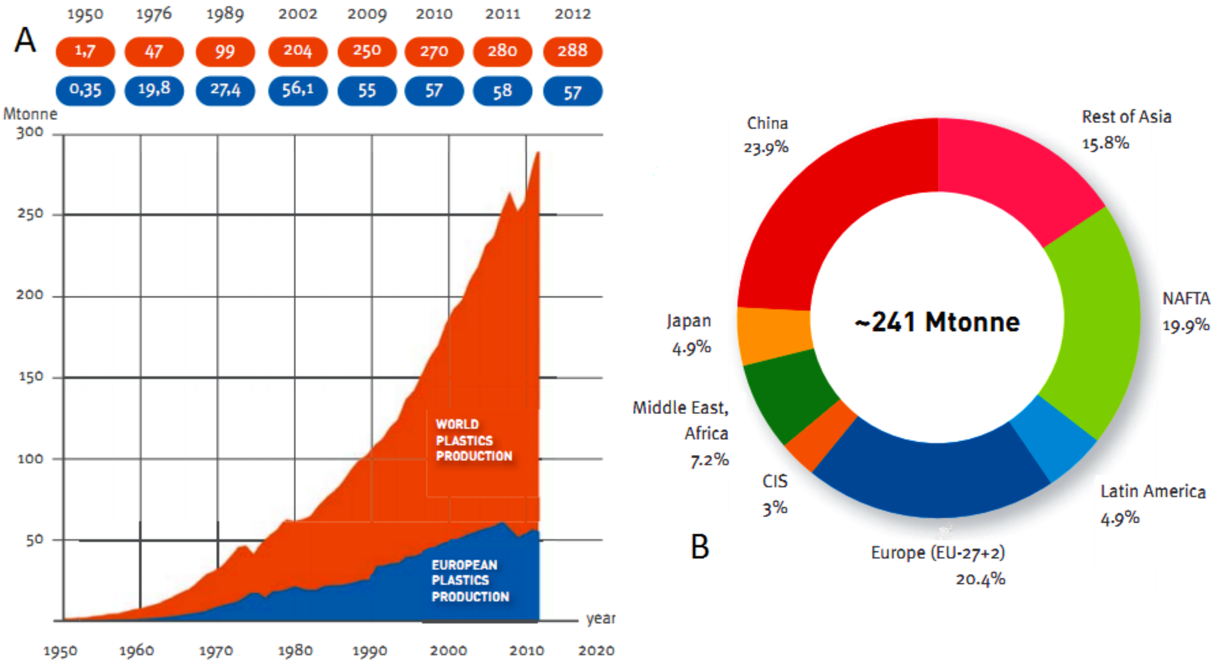

Figure 1.5.4. A: World plastics production 1950-2012; B: World plastics materials production in 2012
(Adopted from Plastics Europe, 2013)

Not long after the commencement of the mass production of plastics, their first negative impacts on the environment were documented. In the 1950s and 1960s plastic spherules were reported in the gastrointestinal tracts of seabirds and fish, as well as the occurrence of plastic-related chemicals in animal tissue (Carpenter *et al.*, 1972; Rothstein, 1972, Waldichuk, 1978; Harper and Fowler, 1987). Accumulation of plastics in the environment is directly linked to their properties, most of all their being inexpensive, making them readily available (Barnes and Milner, 2005, Barnes *et al.*, 2009). The convenience of disposable and cheap plastic items was advertised in the magazines of the time, along with the new, easier way of living (Fig. 1.5.5). Throw-away lifestyle and overconsumption have unquestionably greatly contributed towards marine debris pollution.

Fig. 1.5.5. Image from Life Magazine article '*Throwaway living*' (1955, p. 43).

Recently, Rochman et al. (2013d), proposed classification of plastic waste as hazardous waste due to its associated toxicity and other types of harm to humans and other biota. Types of plastic which they consider should not be produced anymore include: PVC, PS, PUR and polycarbonate (PC). In response to this concern, in November 2013, EPA Region 9 launched a first-time Superfund assessment to further investigate the potential adverse effects of post-consumer plastics on wildlife (Inside EPA, 2014).

### *1.5.3. Microplastics*

Marine debris pollution is a very ‘visual’ pollution, usually accompanied with visually disturbing images of impacted wildlife and littered wilderness, however it seems there might be greater concerns associated with the invisible plastic pollution – microplastics (Thompson *et al.*, 2004; Andrady 2011, Cole, 2011). It has been stressed that more investigation is needed to understand the impacts of microplastics on marine organisms and food web (Wright *et al.*, 2013). Furthermore, marine microplastics are far more difficult to trace back to its origin than macro-debris (Moore, 2008), which presents an additional obstacle in managing this type of pollution.

There are recent reviews on microplastics in the marine environment (Andrady, 2011; Cole, 2011) and their physical impacts on marine organisms (Wright *et al.*, 2013). The definitions of microplastics by different authors include various size-ranges (Andrady, 2011; Cole, 2011), from <10mm (Graham and Thompson, 2009), < 5mm (Barnes *et al.*, 2009), 2-6mm (Derraik, 2002), < 2mm (Ryan *et al.*, 2009) to < 1mm in diameter (Browne *et al.*, 2007; Browne *et al.*, 2010; Claessens *et al.*, 2011). Gregory and Andrady (2003) separate virgin pellets from other microplastatics placing them into a ‘mesolitter’ category. Cole (2011) emphasizes the need for standard size limits of microplastics for valid comparisons between studies.

Cole (2011) discerned two categories of microplastic: primary and secondary. Primary microplastics include all plastic particles designed and manufactured as microscopic objects, such as abrasive microparticles in cosmetics and detergents (e.g. scrubbers) (Zitko and Hanlon, 1991, Fendall and Sewell, 2009), airblast cleaning media (Gregory, 1996) and medical tool for drug delivery (Patel *et al.*, 2009). Abrasive plastic particles, so called ‘micro-beads’ or ‘micro-exfoliates’, in hand and facial cleaners have been in use since the 1980s and have replaced natural abrasives such as ground almond or pumice (Cole, 2011). These microplastics enter the marine environment via wastewater as they are not retained by the wastewater treatments. Fendall and Sewell (2009) analysed 4 water-based facial cleaners purchased in Auckland, New Zealand. The cleaners contained microparticles ranging in average size from ~200-375µm (i.e. 0.2-0.375mm), the sizes easily ingested by plankton.

Secondary microplastics are formed by breakdown of larger plastic items (Thompson *et al.*, 2004, Cole, 2011), which, when exposed to weathering, become embrittled and degrade into smaller pieces (Andrady, 2003). Secondary microplastics may also form by biodegradation of biodegradable component of composite plastics, leaving behind the non-biodegradable plastic particles (Klemchuk, 1990). Additionally, Browne *et al.* (2011) examined the sediments, sewage and washing machine effluent and found that a single garment can release more than 1900 fibers per wash, suggesting that a large proportion of fibers in the environment originate from synthetic clothes which release micro-fibres during washing.

## 1.6. Distribution and quantities of plastics in the marine environment

Reliable estimates of marine debris input in the oceans are, if not impossible (Derraik, 2002), at best, difficult to calculate (Kershaw *et al.*, 2011), because the sources of marine debris pollution are non-point and the pollution is spread globally (Gregory and Andrady, 2003; Ryan *et al.*, 2009). However, further accumulation and expansion of plastics in the environment is inevitable. Firstly, due to universal use of plastics, they have become pervasive among synthetic materials and their production steadily increases each year (Plastics Europe, 2013). Accordingly, plastics increase as a waste component as well, and some of these do not reach landfills and recycling facilities, but accumulate in the environment. Secondly, physical and chemical properties of plastics, which make them durable and non-biodegradable, are responsible for their persistence and accumulation in the environment (Gregory and Andrady 2003; Moore, 2008; Andrady, 2011).

Although plastic pollution is most conspicuous along coastal areas, only 30% of all marine debris floats and ultimately strands on coasts. The majority (70%) is spread over the seafloor (UNEP, 2005).

### *1.6.1. Ocean currents*

The oceans are in constant motion, driven by the Sun's energy and the Earth's rotation. There are two main types of ocean circulation: wind-driven surface currents and density-driven deep-ocean circulation (Brown *et al.*, 2001).

#### *1.6.1.1. Surface currents*

The ocean and the atmosphere are closely linked in one dynamic system. The atmosphere and wind system drive the ocean's circulation and the ocean influences the circulation of the atmosphere through heat transfer. The complex system of some of the ocean processes is depicted in Fig. 1.6.1.

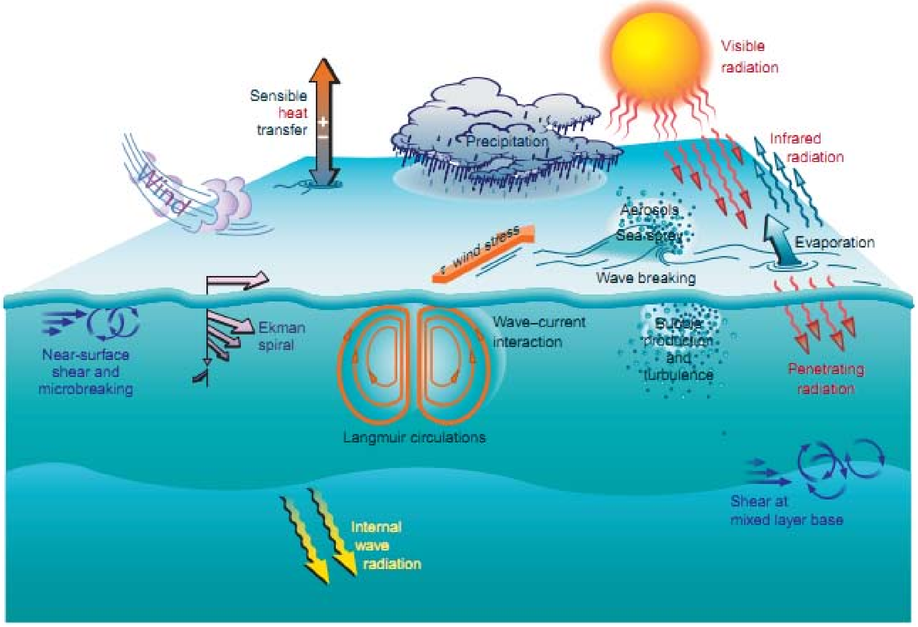

Fig 1.6.1. Schematic drawing of wind- and buoyancy-forced upper-ocean processes.
Courtesy Jayne Doucette, Woods Hole Oceanographic Institution. (adopted from Thorpe (2009)).

The Sun's energy causes the circulation of the atmosphere resulting in winds. At the equator, hot air rises and it is replaced by cooler air from higher latitudes to even out the difference in air density (i.e. pressure). These moving air masses - winds - are deflected clockwise in the northern, and counter-clockwise in the southern hemisphere as a result of Earth's eastward rotation (Fig 1.6.2.). In general, objects moving on the surface of a rotating sphere, due to its curvature and consequent decreasing rotational speed from the equator towards the poles, appear to move in a curve. This apparent curved motion is called the Coriolis effect, to which all objects moving in a rotational system are subjected to (Brown *et al.*, 2001; Stewart, 2008).

When the wind blows over the ocean, the air mass 'pushes' the surface layer of the water through the friction at the interface. This causes surface circulation of the water which reaches down to about one kilometer of depth (Brown *et al.*, 2001; Stewart, 2008).

Simplified, the energy transferred from the wind to the water generates surface gravity waves and drives currents. The frictional force of the wind upon the sea-surface is called wind stress. The greater the speed of the wind, the stronger the wind stress and, consequently, the current generated. The energy of the wind propagates downwards in the water column as a result of internal friction. The friction causes turbulent movement of the water and is characterised by eddy viscosity, which is much greater than the friction in the water moving in a laminar manner, characterised by molecular viscosity. Turbulent eddies transfer the energy to deeper layers of the ocean (Brown *et al.*, 2001).

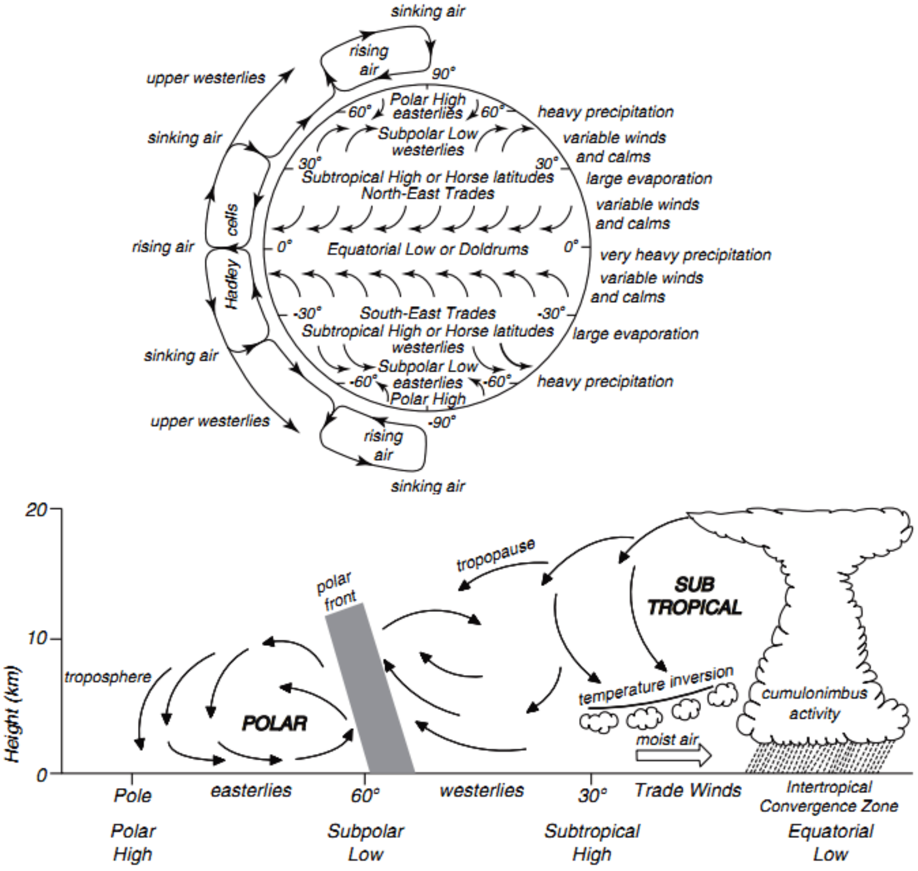

Fig 1.6.2. Global wind system (adopted from Stewart (2008)).

In the 1890s, a Norwegian scientist and explorer, Fridjof Nansen, observed that when the wind blew at sea in the Arctic, the floating ice was deflected from the wind direction by 20-40 degrees to the right. In 1905, V. W. Ekman used his observations to describe the mechanism behind wind-driven currents. Briefly, as mentioned, the wind stress spreads downward in the water column by eddy viscosity and the friction between layers of water. Because all moving objects on the Earth are affected by Coriolis pseudo-force, caused by Earth's rotation, the layers of the water pushed by the wind move in a spiral pattern known as the Ekman spiral (Fig. 1.6.1). The surface layer of the ocean water, under the influence of the wind, is called Ekman layer. There is a similar layer on the bottom of the ocean created by the friction between the sea floor and the bottom water – the bottom Ekman layer (Stewart, 2008). In the global ocean circulation system (Fig. 1.6.3), currents generally deflect to the right in the northern and to the left in the southern hemisphere (Brown *et al.*, 2001).

Major current systems, forming large-scale 'vortices', are also called ocean gyres. In the centre of the gyres water moves vertically, depending on the type of the wind circulation over the ocean. Cyclonic low pressure winds cause divergence of the surface water and upwelling of the cooler water from beneath (i.e. Ekman pumping), while anticyclones cause convergence and sinking ('downwelling') of the surface water in the middle of the gyre. There are two subtropical gyres in the Pacific Ocean, two in the Atlantic and one in the Indian Ocean. Cyclonic gyres are driven by subpolar low pressure systems and are called subpolar gyres (Browne *et al.*, 2001).

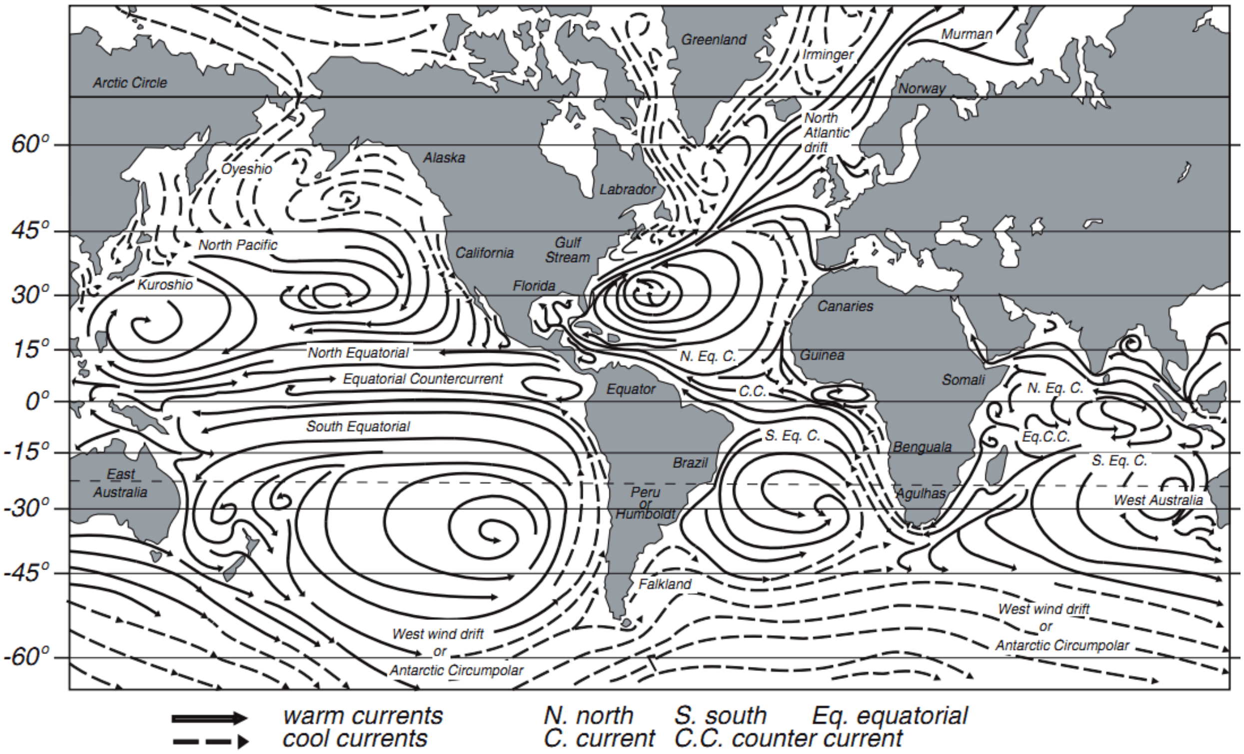

Fig. 1.6.3. 'The time-averaged, surface circulation of the ocean during northern hemisphere winter deduced from a century of oceanographic expeditions. After Tolmazin (1985: 16).' (adopted from Stewart (2008), p. 16).

Observation of oceanic systems is based on both satellite and in situ observations. In situ measurements usually include use of Lagrangian (free drifters) and Eulerian methods (moored instruments). Lagrangian methods use untracked or tracked floating objects. In the 19th and early 20th century, a common practice was to use ‘drift bottles’ and deduce their current interaction by the time and place they were released and found. Sometimes untracked drifters (i.e. plastic drift cards) are still in use as a cheap solution (Brown *et al.*, 2001), and sometimes unconventional methods are used as well. For example, Ebbesmeyer and Ingraham (1994) used drifting marine debris for ocean current modeling. In 1992, a shipping vessel lost a dozen containers at sea due to heavy weather. One of the containers, holding 29,000 plastic bath tub toys, broke in the ocean and the content was released in the northern Pacific Ocean. Ten months later the toys were recorded on the beaches in Alaska. Ebbesmeyer and Ingraham simulated the toys’ drift using a computer program, the Ocean Surface Current Simulations (OSCURS) numerical model.

Other, more conventional Lagrangian methods use satellite tracked drifting floats, such as ARGO floats of the Global Ocean Observing System (GOOS) (Gould *et al.*, 2004) or NOAA’s drifters of the Global Drifter Program (Lumpkin *et al.*, 2013), equipped to record different water properties (e.g. salinity, temperature, velocity). ARGO floats record data from within 2000m of the water column, while the drifters float at the surface.

#### *1.6.1.2. Deep-ocean circulation*

The Sun’s energy also influences the density of the surface waters through changes in temperature and salinity. Warm surface water reaching higher latitudes is cooled down by cold winds, and as its density is increased by formation of ice, it becomes heavier and it gradually sinks to the bottom. Density-driven circulation, caused by sinking of cooler and saltier layers of the water to the bottom in polar regions, is often called termohaline or abyssal circulation (Brown *et al.*, 2001).

### *1.6.2. Distribution pathways*

Plastics started accumulating in the oceans in the 1950s, following economic and human population growth (Kirkley and McConnell, 1997; Thompson *et al.*, 2009a). Originating from land and ocean activities, plastic items enter the ocean mainly through waterways, by wind transport and through loss from boats. Plastics which are negatively buoyant sink to the bottom and the positively buoyant ones are carried away by winds and currents, sometimes far from the source of input (UNEP, 2005). Floating plastics either accumulate on the shores of continents and islands or gather in oceans' gyres (Derraik, 2002). Ryan *et al.* (2009) summarised methods of studying and monitoring marine debris pollution, as well as listed the sources and described the fate of debris in the marine environment (Fig. 1.6.4.).

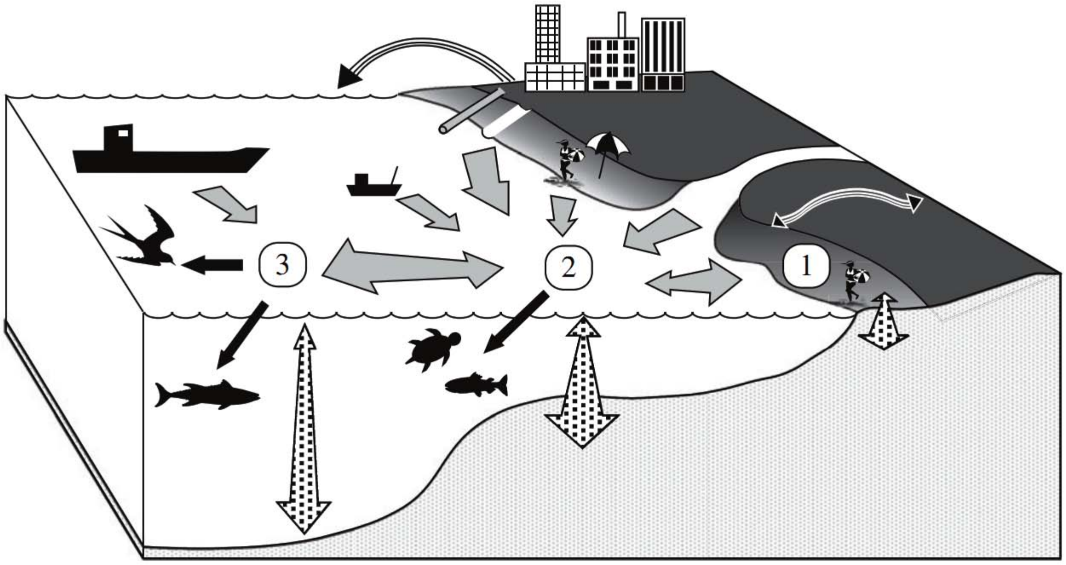

Fig. 1.6.4. Schematic diagram of the main marine debris sources and their pathway in the environment (adopted from Ryan *et al.*, 2009). Sinks: 1) beaches, 2) coastal waters and sediments and 3) open ocean. (curved arrows depict wind-blown litter, grey arrows water-borne litter, stippled arrows vertical movement through the water column (inc. burial in the sediment) and black arrows ingestion by marine organisms.

Plastic debris can be broadly divided into 3 size groups: macro-debris (>20mm), meso-debris (5–20 mm) and micro-debris (<5mm) (Ryan *et al.*, 2009). However, other size limits have been used as well (e.g. Gregory, 1999b). Since there is still no widely accepted consensus on sampling standard, size limits are usually variable and defined by authors for each study differently.

#### *1.6.2.1. Surface debris and 'garbage patches'*

Distribution of floating debris in the ocean depends on its mass, buoyancy and persistence of the material from which it is made (Moore *et al.*, 2001). Distribution is also patchy due to uneven input, as well as winds and ocean currents which disperse it (Shaw and Mapes, 1979; Pruter, 1987). Convergent zones concentrate marine debris, while divergent disperse it. The patterns of marine debris dispersion by ocean circulation are being studied using different current models, such as 'Ocean Surface Current analysis –

Realtime' (OSCAR) (Martinez *et al.*, 2009) and 'Hybrid Coordinate Ocean Model' (HYCOM) (Lebreton *et al.*, 2012). These and other current models are described in more detail by Potemra (2012).

The assessment of the quantities of floating debris generally includes two different methods: collection and visual observation. Debris collection is usually performed by towing a neuston net, most commonly with the 0.33mm mesh opening, along the surface of the water and collecting floating debris in the cod end (e.g. Wilber, 1987; Ryan, 1988; Moore *et al.*, 2005; Ryan *et al.*, 2009; Eriksen *et al.*, 2013; Goldstein *et al.*, 2013a; Collignon *et al.*, 2014). This type of assessment is used for studying smaller debris and microplastics. Visual observations are used to assess the quantities of larger floating marine debris. These include ship-based sighting surveys (e.g. Thiel *et al.*, 2011; Goldstein *et al.*, 2013a; Ryan, 2014) and aerial surveys by planes (e.g. Lecke-Mitchell and Mullin, 1997; Pichel *et al.*, 2007) or various airborne sensors, including small unmanned airborne vehicles (SUAV) (Veenstra and Churnside, 2012). The results from studies using different methodologies vary considerably and, due to different debris sizes targeted, are not comparable. Table 1.6.1. lists studies of surface marine debris by (i) visual observations and (ii) sample collections, which are reported in two most commonly used measurement units a) item km$^{-2}$ and b) item m$^{-3}$. Note that some values given in the text were recalculated from the units provided in the paper into the commonly used units in the table (e.g. item 100m$^{-2}$ into item km$^{-2}$).

Early encounters with plastic pollution of surface waters were reported in the 1970s from the Atlantic and Pacific oceans. In 1969 and 1970, Heyerdahl and his crew observed severe marine pollution by floating tar and plastic on their experimental voyage on papyrus rafts across the Atlantic, from Africa to South America (Heyerdahl, 1971). Their reports were only descriptive. Shortly thereafter, Carpenter and Smith (1972) reported the quantities of plastic spherules (2.5-5mm) collected in 11 neuston tows from the Sargasso Sea with average densities of about 3,500 pieces km$^{-2}$ and a maximum of 12,080 pieces km$^{-2}$.

The first documented observations of floating debris in Pacific waters in 1972 were published by Venrick *et al.* (1973). They estimated that one plastic bottle can be found afloat every 2km$^2$, which, when extrapolated to the whole North Pacific, totals 35,400,400 bottles. However, they did not take into account the fact that they were sampling in the convergence zone and that the distribution of floating debris is uneven across the ocean surface. The first surface trawls carried out across the north Pacific waters, from Tokyo to British Columbia, resulted in densities greater than recorded in the Atlantic (Wong *et al.*, 1974). The greatest density of floating tar was found in the Northwest Pacific, while plastics were more abundant the Northeast Pacific (around 35N, 143E), with the maximum density of 34,000 item km$^{-2}$. Interestingly, Day *et al.* (1990) found greater debris densities in the central and western parts of the Pacific, than in the eastern.

As mentioned earlier, in each ocean, the currents gather floating plastic, sourced from maritime and land-based human activities, in convergent zones. A well known convergence zone in the Northeast Atlantic, the so-called Sargasso Sea or the centre of the North Atlantic gyre, is located off the east coast of US (Carpenter and Smith, 1972). Floating material in the Northern Pacific Ocean concentrates in two areas, east of Japan and northeast of Hawaii (Howell *et al.*, 2012). The northeastern Pacific subtropical gyre was popularised by Charles Moore and the term 'Great Pacific Garbage Patch' was coined by the oceanographer Curtis Ebbesmeyer in 1997 and is still in use colloquially (Moore, 2003; Kaiser, 2010). Greater debris concentrations were also recently found in the South Pacific, between Pitcairn Islands and South America (Eriksen *et al.*, 2013) and in the South Atlantic half way between Cape Town and the island Tristan da Cunha (Ryan, 2014). Potentially (or expectedly) there is a fifth 'garbage patch' in the Indian Ocean as well.

> *'A piece of plastic found in an albatross stomach last year bore a serial number that was traced to a World War II seaplane shot down in 1944. Computer models re-creating the object's odyssey*

> *showed it spent a decade in a gyre known as the Western Garbage Patch, just south of Japan, and then drifted 6,000 miles to the Eastern Garbage Patch off the West Coast of the U.S., where it spun in circles for the next 50 years.'* – Kenneth R. Weiss, L.A. Times, 2 August 2006

Temporal trends in surface plastic pollution in the Sargasso Sea have been studied by the Sea Education Association (Law *et al.*, 2010), and in the Southern California current system by the Scripps Institution of Oceanography (Gilfillan *et al.*, 2009). The 22 year sample collection from the Sargasso Sea does not show an increase in plastic content over time despite the evident increase in plastic production and discard rates (Law *et al.*, 2010). Similarly, the samples from 1984, 1994 and 2007 from the waters off Southern California did not reveal an increasing trend either. The maximum recorded density was in 2007, however, the greatest average density was recorded in 1994 (Gilfillan *et al.*, 2009). Ogi and Fukumoto (2000) reported initial increase in the 1970-80s and subsequent ten-fold decrease in the 1990s in Japanese coastal waters. However, exceptionally high densities were recorded in the recent years in the Northeast Pacific subtropical gyre (Goldstein *et al.*, 2013a). In general, Pacific waters appear to be more polluted by floating plastic than the Atlantic and Indian Ocean (Table 1.6.1.).
Assessments of small and micro-debris (<5mm) result in considerably greater densities than the assessments of larger debris. Dufault and Whitehead (1994) studied both small and large debris off Nova Scotia, close to Sable Island, employing both observation and collection methods. The densities of large debris around the island were 31.6 items km$^{-2}$, while small debris was present in much greater density - 84,000 items km$^{-2}$ on average. Goldstein *et al.*, (2013a) found generally much higher densities in the Northeast Pacific with a considerable difference between large and small debris counts, 1400-3200 item km$^{-2}$ and 448,000 item km$^{-2}$, respectively (Table 1.6.1.).

Moret-Ferguson *et al.* (2010) analysed physical properties of floating plastics from almost 750 surface tow samples collected from Cape Cod to the Caribbean Sea between 1991 and 2007. They found that more than 88% of debris were fragments smaller than 10mm and 95% were lighter than 0.05g. Great majority (99%) of plastics had densities lower than the surface seawater in the Northwest Atlantic (1.025g/mL) and ranged from 0.808 to 1.238 g/mL; 47% of the samples had densities between 0.97 and 1.04, the density of high-density polyethylene (HDPE, 0.89-0.93 g/mL) and polystyrene (PS, 1.04-1.08 g/mL). They also collected beached plastics and measured their density and found that the majority was in the group of low-density polyethylene (LDPE, 0.89-0.93 g/mL). The authors suggested that the processes at sea, such as weathering and biofouling, potentially change the density of floating plastics.

#### *1.6.2.2. Submerged debris in the water column*

Plastic debris floats on the surface of the water, and also throughout the water column due to the variable densities of different plastic materials. Debris can also change in position in the water column due to biofouling, which makes debris items negatively buoyant (Ye and Andrady, 1991; Lobelle and Cunliffe, 2011). Some studies showed that plastics may also undergo rapid defouling and resurface (Ye and Andrady 1991). Furthermore, Kukulka *et al.* (2012) noted that the wind influences the vertical distribution of debris as well and that surface sampling alone can underestimate the quantity of debris throughout the water column.

The reports show that the contamination of subsurface waters by plastic particles is usually less severe than that of surface waters. Moore *et al.* (2005) investigated the relationship between microdebris densities and rain events and found that the surface layer and 5 m below the surface the water contains considerably more plastics after rainfalls than before. Conversely, the microplastics were more abundant in the near bottom water (at 30m depth) before the rain event than afterwards, which indicates the potential resuspension of the particles in bottom layer as a result of the rain even and associated environmental factors. However, subsurface microplastic densities are still much lower than the surface and near-bottom densities (Moore *et al.*, 2005; Gilfillan *et al.*, 2009). The studies which dealt with subsurface plastic contamination are listed in the Table 1.6.2. Note that some values were recalculated from the original.

### *1.6.2.3. Shore accumulations- surface and buried debris*

Beach litter reflects the state of near shore and offshore waters and associated anthropogenic maritime activities, as well as activities held on the beach and nearby areas. Beach debris studies are the most common way of studying marine debris pollution (Ryan *et al.*, 2009). The assessment of the abundance, type, origin and accumulation rate of the beach debris is a practical, cost-effective method used to define which human activities contribute towards marine pollution in a certain area (Ribic and Johnson, 1990) and, consequently, to propose the effective measures to improve marine debris management.

Table 1.6.3. (a, b) lists studies of stranded marine debris along the coasts. It does not include the publications where it was significantly time-consuming, difficult or impossible to extract the necessary information from (debris densities) (e.g. Walker *et al.*, 2006) or the units were incorrect (e.g. Zhou *et al.*, 2011). The two most commonly used units are item $m^{-1}$ (a) and item $m^{-2}$ (b), thus the results which expressed the densities only as weight (per area or length, or standing stock) or number of pieces (standing stock) were also not included in the table. The studies of beach debris are the least standardised of all marine debris studies, which causes many difficulties in comparative studies and meta-analyses. The densities from the Table 1.6.3. a and b, albeit expressed in the same units, are not entirely comparable due to, among others, different size limits of targeted debris, variable debris categories and the method of averaging the densities (e.g. temporally pooled).

Most studies of coastal debris are either short-term or one-off studies. There are several long-term studies which investigated the trends over a decade or more (Hayward, 1999; Edyvanne *et al.*, 2004; Eglinton *et al.*, 2006; Morishige *et al.*, 2007; Ribic *et al.*, 2010; Ribic *et al.*, 2012b). Morishige *et al.* (2007) studied beach debris accumulation, in relation to El Niño and La Niña events, on Tern Island, French Frigate shoals in Hawaiian Islands, over 16 years (1990-2006). They found that debris deposition is significantly greater during El Niño period. They attributed this phenomenon to the winter southward shift of the subtropical convergence zone which is more pronounced during El Niño events, because the convergence zone, which concentrates marine debris, approaches northwest Hawaiian beaches. Significantly elevated debris quantities were recorded from 2002-04, which were El Niño years. Generally, annual accumulations did not show increasing or decreasing trends over time.

As a part of US National Marine Debris Monitoring Program, Ribic *et al.* (2012b) analysed shore debris data collected from the American Pacific coast (1998-2007) and Hawai'i (2000-2007). Hawai'i had the highest debris loads, while the least debris deposited in the North Pacific coast of America. They also observed a relationship between ENSO cycle and debris loads in Hawai'i. A general decrease over time was observed for most surveyed regions. Conversely, in a long-term study on the western coast of Northern Island of New Zealand, Hayward (1999) reported a clear increase from 1974 to 1997.

Reported densities of marine debris vary substantially locally, regionally and worldwide. For example, Slip and Burton (1991) reported densities of 0.009 item $m^{-1}$ on Macquarie Island, while Gregory (1999b) reported great densities of plastic pellets in Auckland (>100,000 item $m^{-1}$), Wellington (> 40,000 item $m^{-1}$), Tonga (>> 1000 item $m^{-1}$) and Fiji (>> 100 item $m^{-1}$). Considerable differences in this case are primarily due to different sizes of targeted debris (Table 1.6.3a), but are also likely to be related to the proximity to the population centres, different sediment types and morphodynamic conditions.
Bowman *et al.* (1998) noted that several authors suggested a pollution density index to determine the level of marine debris pollution in a certain area such that:

- low pollution index: < 1 item $m^{-1}$
- medium pollution index: 1-10 item $m^{-1}$
- high pollution index: >10-100 item $m^{-1}$

To investigate the 'behaviour' and fate of beach debris, some authors applied mark-recapture experiments (Dixon and Cooke, 1977; Garrity and Levings 1993; Johnson 1989; Williams and Tudor, 2001; Ivar do Sul *et al.*, 2014). Marine debris behaves in the same way as the sediment, and has a source, pathway and sink (Williams and Tudor, 2001; Rios *et al.*, 2010). Dixon and Cooke (1977) found that shore debris retention rates depend on the debris fabrication material and the beach type. Firstly, plastic containers had longer retention periods than the glass ones, as the glass breaks more easily than plastics. Secondly, sandy beaches are wider and have a gentler slope than narrow shingle beaches, thus the overlap between the high water marks on a sandy beach are stretched over a larger area, allowing the debris to remain undisturbed by waves for longer periods than on a shingle beach. The authors further suggested that the main loss mechanisms are wave action, which withdraw debris from the beach back into the ocean, and burial in the sediment. They also found that 6% of the containers were buried and later exhumed by waves. This indicates that the hydrodynamic processes of the beach have a direct influence on marine debris 'behaviour'. A similar experiment was performed by Williams and Tudor (2001) who studied debris burial and exhumation. They found the proportional relationship between the wind speed (i.e. wave energy) and the fresh litter inputs onto the beach comprised of 'new' debris from the ocean and 'old' exhumed debris. An interesting observation of crab burrows trapping beach debris indicated another potential mechanism of debris burial (Iribarne *et al.*, 2000).

Large number of studies, summarized in the Table 1.6.3.c, aimed to estimate the amount of debris buried in shore sediments. As can be seen from the table, the variety of measurement units is exceptional. The most recent analysis of beach sediments in Sao Paulo, Brazil, showed that they contained plastic pellets at depths as deep as 2m (Turra *et al.*, 2014). In the first 20cm of the sediment from Santos Beach, they found the density of 5,385 item $m^{-3}$, which at depth of 2m reached a cumulative value of almost 30,000 pellets per $1m^3$. Kusui and Noda (2003) assessed the ratio of surface and buried debris in the top 5cm of the sand on Japanese and Russian beaches. They reported average surface densities of 3.41 and 0.21 item $m^{-2}$, respectively. The average densities of buried debris were 2.61 and 0.31 item $m^{-2}$, respectively. For most beaches the ratio between buried and surface debris was around 1.

Remarkably high densities of sediment debris were documented on Midway Atoll (441 item $L^{-1}$) (McDermid and McMullen, 2004), Japan (290.5 pieces $L^{-1}$) (Fujieda *et al.*, 2002), Portuguese coast (2421 item $m^{-2}$) (Antunes *et al.*, 2013), Easter Island (805 item $m^{-2}$) (Hidalgo-Ruz and Thiel, 2013) and Kauai Island (96.8 item $m^{-2}$ day $^{-1}$) (Cooper and Corcoran, 2010). It should be noted that Hawaiian Islands and Easter Island lie close to the centres of the convergent zones in the North and South Pacific, respectively.

Many of the studies of buried debris, and some of surface debris, focus on microplastics or the microplastics were the main component of assessed debris (see Table 1.6.3.). Of particular interest in some areas are plastic pellets which sometimes comprise more than half of all sediment debris (Antunes *et al.*, 2013).

### *1.6.2.4. Sea floor debris*

Although majority of marine debris is believed to be spread on the sea floor (70% - UNEP, 2005), little attention has not been given to this compartment of the ocean, presumably due to the costly and time-consuming survey methods. As previously mentioned, seabed marine debris is either composed of negatively buoyant (e.g. metal, glass, PVC, PET) or from positively and neutrally buoyant materials (wood, PP, PE) which become negatively buoyant as a result of biofouling (Holmstrom, 1975; Ye and

Andrady, 1991). Moore (2008) also suggests that neutrally buoyant debris may entrap some sediment and remain submerged on the sea bottom.

Methods for studying marine debris on the sea floor most commonly include trawling (collection), SCUBA diving and snorkeling (visual observation or collection) and image and video analysis (sonar, remotely operated vehicle (ROV), manned and unmanned submersible) (Table 1.6.4.a). Diving and snorkeling are used in shallow survey areas. Trawling technique is employed up to several hundreds of meters, while the deepest ocean areas are surveyed by ROVs and submersibles. Studies which focused on seabed marine debris are listed in the Table 1.6.4.a. Here, as well, were taken into account only studies with results expressed in commonly used measurement units (item $km^{-2}$ or convertible into) and the ones with easily extractable and clearly presented information.

Submerged seabed debris has been documented in shallow seas, reefs and lagoons (e.g. Boland and Donohue, 2003; Richards and Beger, 2011), as well as on the deep ocean floor, canyons and trenches (e.g. Galgani *et al.*, 2000; Keller *et al.*, 2010; Miyake *et al.*, 2011; Mordecai *et al.*, 2011). Distribution of debris across the sea floor depends on the sources, bottom morphology and hydrodynamics (Galgani *et al.*, 2000; Ryan *et al.*, 2009). Greater densities of bottom debris on the continental shelves of heavily populated areas are usually associated with river runoffs (Galgani *et al.*, 2000). Pham *et al.* (2014) found the highest litter densities in the submarine canyons. Offshore and open ocean debris accumulations are usually related to shipping routes and fishing activities (Watters *et al.*, 2010; Pham *et al.*, 2013), but fishing debris is also frequently found in the shallow seas, such as coral reefs of Hawaii (Dameron *et al.*, 2007), coastal areas of Greece (Katsanevakis and Katsarou, 2004) or bays of Alaskan islands (Stevens *et al.*, 2000).

Miyake *et al.* (2011) analysed deep sea video recordings, taken by a manned submersible (deeper than 1000m) and ROVs, from the Japan Agency for Marine-Earth Science and Technology (JAMSTEC). They documented man-made debris off the Japanese coast as deep as 7216m (a waste can). Debris accumulates in deep ocean traps, such as a gap in the Japan Trench, off Sanriku, made by an earthquake. Schlining *et al.*, 2013 analysed video records taken by ROVs in Monterey Bay, California, up to almost 4000m, from 1989 to 2011. Similarly to reports from Pham *et al.* (2014), they also found debris to accumulate in submarine canyons together with natural debris. This indicates that underwater debris is subjected to the same hydrographic conditions as natural materials, which is valuable information for managing seabed debris. The trends in debris abundance over time were not observed. The same area was surveyed on several occasions, from 1993 to 2007, by another team using a manned submersible up to depths of 365m and they found significantly increasing trends in the amount of recreational fishing debris (fishing line) between 1990s and 2007 (Watters *et al.*, 2010).

A small number of surveys, indicated in the Table 1.6.4.b, were completed to examine the microplastic pollution of the subtidal sediments. Thompson *et al.* (2004) collected samples of the sediment from beaches, estuaries and subtidal around Plymouth, UK, and they found significantly more microplastic content in subtidal sediments. Van Cauwenberghe *et al.* (2013) collected deep-sea sediment samples from several locations in the Atlantic ocean floor (Porcupine Abyssal Plain – 4844m, Congo Canyon – 4785m, Southern Ocean – 4881m) and Nile Deep Sea Fan in the Mediterranean (1176m). The microplastic particles were found in all locations except the Congo Canyon. This was the first evidence of microplastic pollution reaching the abyssal depths of oceans.

## 1.7. Consequences of marine debris pollution

The number of reports on the impacts of marine debris pollution on wildlife and humans has increased over time, and mainly originate from North America (117), Australasia (56) and Europe (52) (Fig 1.7.1.). A recent review provided a number of 663 species documented to be affected by marine debris pollution (Secretariat of the Convention on Biological Diversity and the Scientific and Technical Advisory Panel – GEF, 2012), which is a 40% more than previously reported in 1997 (247 species) (Laist, 1997). Most adversely affected taxonomic groups are sea birds (161) and fish (114). A total number of 47,963 individuals and 373 species were reported to have either ingested or became entangled in marine debris. Plastic is the dominant material in most interactions with wildlife, reported in 76.5% of all publications (319) (Fig. 1.7.2).

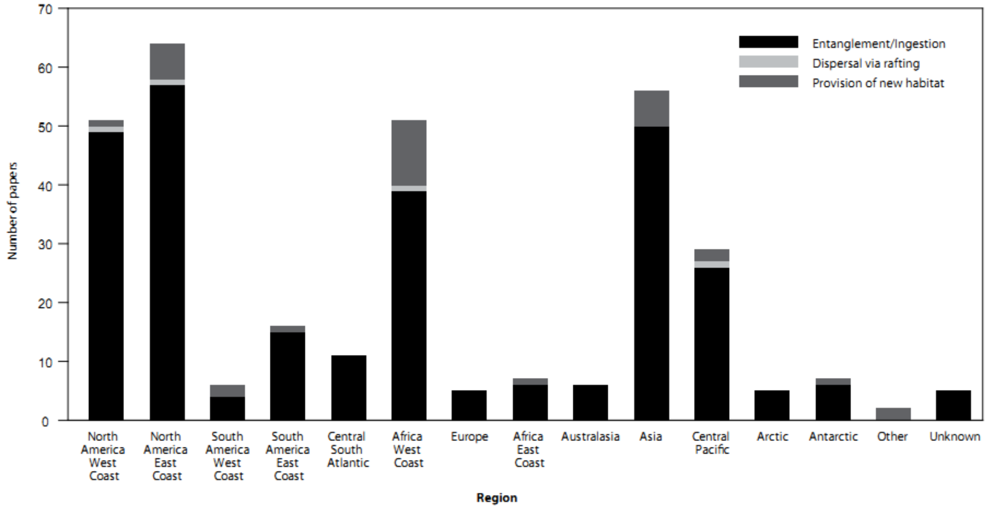

Fig. 1.7.1. Number of reports on marine debris impacts on wildlife according to different regions (Other – studies included multiple regions, Unknown – studies did not report location). (Secretariat of the Convention on Biological Diversity and the Scientific and Technical Advisory Panel – GEF, 2012).

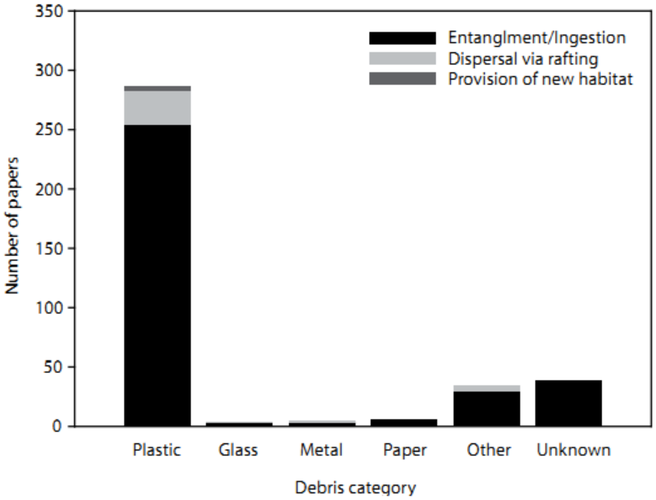

Fig. 1.7.2. Number of papers reporting various impacts of different marine debris categories. (Secretariat of the Convention on Biological Diversity and the Scientific and Technical Advisory Panel – GEF, 2012)

### *1.7.1. Plastic ingestion*

#### *1.7.1.1. Seabirds*

Reports on plastic ingestion by seabirds were among the first reports on the impacts of marine debris on marine organisms. In the 1950s, Harper and Folwer (1987) found plastic pellets in beached prions (*Pachyptila salvini*) and (*P. desolata*) in New Zealand. In 1962, Rothstein (1973) reported plastic particles in Leach's Petrels (*Oceanodroma leucorhoa*) collected in Newfoundland. In 1971, Hays and Cormons (1974) found plastic spherules in the pellets of gulls and terns on Great Gull Island, New York. Connors and Smith (1982) collected 7 dead Red Phalaropes (*Phalaropus fulicarius*) in 1980 from Botega Bay, California, and 6 of them contained up to 14 plastic pieces, 6.7 on average. Van Franeker (1985) collected 65 dead seabirds Northern Atlantic Fulmars (*Fulmarus glacislis*), which still contained intact stomachs, from 1982-84 from Dutch coast, and plastic particles were found in 92% of birds, on average 12 pieces per bird, with a maximum of 96 fragments. These and later reports demonstrate an increase in the occurrence of plastic ingestion since the 1970s (Table 1.7.1.).

Obtaining the data on plastic ingestion by seabirds has been completed in several ways. Generally, there are lethal and non-lethal methods. The former usually includes shooting the birds (e.g. Spear *et al.*, 1995, Vlietstra and Parga, 2002), or euthanizing the injured birds (e.g. Auman *et al.*, 1997). Non-lethal methods include collecting dead birds, either stranded (e.g. Acampora *et al.*, 2014), or as fisheries bycatch (e.g. Blight and Burger, 1997; Gray *et al.*, 2012), or capturing live birds (e.g. Sileo *et al.*, 1990; Hutton *et al.*, 2008). Ingested plastic is recovered from live birds by induced emsesis by stomach flushing. Even though being proven an effective method (Bond and Lavers, 2013), some studies showed it is not 100% accurate and the reguritated quantities of ingested plastic should be considered underestimates of total body burdens (Lavers *et al.*, 2014).

From 312 known seabird species, 119 were reported to ingest plastic debris (Secretariat of the Convention on Biological Diversity and the Scientific and Technical Advisory Panel – GEF, 2012). Of these, the species most commonly reported, and prone to plastic ingestion more than others, belong to the order Procellariiformes and include albatrosses, petrels, fulmars and shearwaters (Azzarello and van Vleet 1987; Ryan, 1987). Spear *et al.* (1995) examined the incidence of plastic in different orders of seabirds and found that the proportion of ingestion by Procellariiformes was 71% while by Charadriiformes and Pelecaniformes only 27%. Most Procellariiformes species have small gizzards and rarely regurgitate indigestible content, which is the likely cause of their high vulnerability. Most frequently reported species in the Pacific affected by ingestion are albatrosses, particularly Laysan (*Phoebastria immutabilis*) and Black-footed Albatrosses (*Phoebastria nigripes*) (Sileo *et al.*, 1990; Auman *et al.*, 1997; Gray *et al.*, 2012) and shearwaters, especially Flesh-footed (*Puffinus carneipes*) (Hutton *et al.*, 2008; Lavers *et al.*, 2014), Wedge-tailed (*Puffinus pacificus*) (Verlis *et al.*, 2013) and Short-tailed shearwaters (*Puffinus tenuirostris*) (Acampora *et al.*, 2013). Pacific Northern Fulmars (*Fulmarus glacialis*) (Mallory 2008; Avery-Gomm *et al.*, 2012) and some petrels (Blight and Burger, 1997) have also been documented to ingest large quantities of plastic. Sileo *et al.* (1990) noted that birds least affected by plastic ingestion were terns and noddies, while Avery *et al.* (2013) observed no evidence of plastic ingestion by Rhinoceros Auklet, Marbled Murrelet, Ancient Murrelet or Pigeon Guillemot, and only low levels in Common Murre. Laysan and Black-footed Albatross are listed as near threatened under the IUCN Red List, while shearwaters and Northern Fulmars are of lesser concern.

Buxton *et al.* (2013) surveyed several islands located in New Zealand coastal waters of the North Island with seabird colonies to examine the existence of plastic fragments most likely related to ingestion and regurgitation. They found the largest densities of plastics on Ohinau Island where the most numerous seabird species was the Flesh-footed Shearwater. This species is more susceptible to plastic ingestion than other shearwaters (Hutton *et al.*, 2008).

Hays and Cormons (1974) suggested that there are two possible ways of ingesting plastic spherules: directly, by ingesting the spherules, or indirectly, by secondary ingestion, or eating small fish which ingested the spherules. Sileo *et al.* (1990) found plastic ingestion to be more prevalent in seabirds which fed at the sea surface. Eighty percent of floating debris, mainly Styrofoam and spongious plastic, stranded on the Dutch coast exhibited peckmarks made by birds at sea (Cadee, 2002). Avery-Gomm *et al.* (2012) reported that 95.7% of plastic ingested by northern fulmars stranded on the beaches of the west coast of Canada and US were mainly 'user' plastic (as opposed to industrial plastic pellets), such as twine, rope, fishing line, Styrofoam, fiber sponge, food wrap (sheeting) and hard plastic pieces (e.g. bottle tops). In addition to eating plastic, seabirds sometimes swallow sinkers and other fishing gear, which can be particularly harmful due to potential lead poisoning (Franson *et al.*, 2003; Phillips *et al.*, 2010).

Another mode of plastic intake in seabirds occurs through parental delivery (Carey, 2011; Rodriguez *et al.*, 2012). Adult seabirds collect plastic pieces at sea, together with food items, and bring them to feed the fledging chicks. Acampora *et al.* (2014) found more plastic in the chicks than adult birds.

The negative effects of plastic on seabirds and a potential pathway of affecting the seabird welfare were illustrated by Lavers *et al.* (2014) (Fig. 1.7.3.). In the early studies, the authors did not encounter any serious health problems with birds in relation to ingested plastic, however they did stress that, since the plastic is entering the food web and may cause trauma due to satiation and mechanical blockages, further investigation is needed into the issue (Hays and Cormons, 1974; Auman *et al.*, 1997).

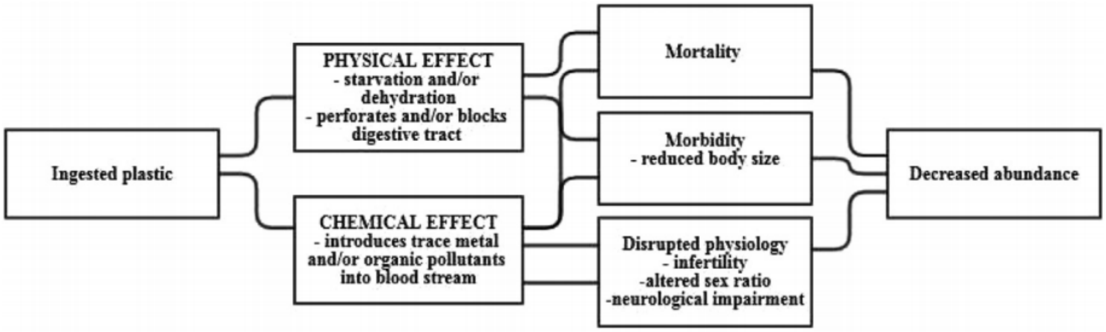

Fig. 1.7.3. Potential pathway through which ingested plastic can affect marine wildlife (adopted from Lavers *et al.*, 2014).

Ingested plastic may have lethal and sublethal effects on sea birds, including physiological and chemical effects. Physiological effects include obstruction of the gastro-intestinal tract, followed by inability to feed properly and subsequent starvation, ulceration of the stomach lining or perforation. Plastic ingestion may also lead to blockage of gastric enzyme secretion causing decreased feeding stimulus and fat deposition, and hormonal changes related to reproduction (Connors and Smith, 1982; Azzarello and van Vleet, 1987; Ryan, 1990). Pierce *et al.*, (2004) performed a necropsy on two birds, a Northern Gannet (*Morus bassanus*) and a Greater Shearwater (*Puffinus gravis*), which died in rehabilitation clinics in Massachusetts. Both birds seem to have died from only 1 piece of plastic in their gizzards.

Plastic materials readily adsorb chemicals from the surrounding water and, once ingested, the chemicals are released from the plastic and deposited in animal's tissue. Plastics may also leach the additives, chemicals added during manufacturing of plastics. Ryan *et al.* (1988) correlated the amounts of ingested plastic with organochlorines (PCBS, DDEs, DDTs and dieldrin) in the fat tissue and eggs of greater shearwaters and found positive correlation only for PCBs. They assumed that PCBs leached from the ingested plastic. Similar findings were obtained in a study on shearwaters and their prey collected from

North Pacific waters. The authors found flame retardants (additives in plastic) in the tissue of seabirds and plastic, but not their prey, indicating that the chemicals were released from ingested plastic (Tanaka *et al.*, 2013).

Lavers *et al.* (2014) examined the relationship between the quantity of ingested plastic and body condition, as well as levels of trace elements in Flesh-footed Shearwaters (*P. carneipes*). The results showed that fledglings with increased plastic loads displayed underdeveloped body features, such as reduced body mass, wing chord and head bill length. Increased levels of chromium and silver were found to be positively correlated with plastic load. Increased chromium may be related to neurotoxic effects in seabirds, while the adverse effects of silver are still poorly understood.

Reports on temporal trends of plastic ingestion in seabirds either show no changes or increase. Ryan (2008) did not find changes in the quantity of ingested plastic, but there was a change in the composition over time, a 44-79% decrease in the proportion of ingested plastic pellets from 1980s to 2000s in the South Atlantic and western Indian Ocean. Van Franeker *et al.* (2011) also reported a decrease in industrial (plastic pellets), but increase of ‘user’ plastic since the 1980s in the Netherlands. Lavers *et al.* (2014) reported an increase in plastic ingestion from 79% of sampled Flesh-footed Shearwaters in 2005-07 to 90% in 2011 at Lord Howe Island, Australia. Reid *et al.* (2013) proposed plastic ingestion as a potential cause of a 19% shearwater population decline from 1978 to 2002, along with road-kills and fisheries by-catch.

Seabirds, as top predators, can be used as indicators of the marine ecosystem health conditions (Mallory *et al.*, 2010; Lavers *et al.*, 2014). Van Franeker *et al.* (2012) and Avery-Gomm *et al.* (2012) propose Northern Fulmars to be used for biomonitoring. A 30-year study on northern fulmars showed that 95% of stranded birds collected from the coast of the North Sea in the Netherlands contained plastic. Between 1980s and 2007, incidence of plastic ingestion increased from 81% of the birds in the 1980s, to 98% from 1999-2003 and declined to 93% from 2003-2007. With respect to the quantities of ingested plastic, the average number per bird increased from 14.6 in the 1980s to a maximum of 33.5 from 1999-2003 and decreased to 26.5 from 2003-2007 (van Franeker *et al.*, 2011). Avery-Gomm *et al.* (2013) also report an increase in plastic ingestion by Northern Fulmars in the North Pacific over the past 40 years.

#### *1.7.1.2. Sea turtles*

The literature on marine debris ingestion by sea turtles, particularly from the Pacific waters, is quite scarce. The issue of plastic ingestion by sea turtles became more prominent in the 1980s (Carr, 1987). Schuyler *et al.* (2013) reviewed the available literature on the matter and analysed 37 studies from 1985 to 2012. In regard to the methodology, they stated that there is no reliable method for assessing plastic ingestion in live turtle populations, thus the studies included in their review are based on the results of necropsy of minimum seven individuals. (Examination of live animals and analyses of esophageal lavage and fecal samples resulted in underestimates of the amounts of ingested debris. Stomach flushing generally yielded less debris than the fecal excretion (Seminoff *et al.*, 2002).

Six of the seven sea turtle species are affected by plastic ingestion (Secretariat of the Convention on Biological Diversity and the Scientific and Technical Advisory Panel – GEF, 2012). According to the findings of Schuyler *et al.* (2013), hawksbill turtles (*Eretmochelys imbricate*) are most likely to ingest debris, while the least likely are Kemp’s ridley turtles (*Lepidochelys kempii*) (Fig. 1.7.4). They also noted that carnivorous species are less likely to ingest debris than herbivorous and gelatinivorous. Five sea turtle species from Fig. 1.7.4. are on the IUCN Red list as endangered or critically endangered: green (*Chelonia mydas*) and loggerhead turtles (*Caretta caretta*) are endangered, and hawksbill, leatherback (*Dermochelys coriacea*) and Kemp’s ridley turtles are critically endangered.

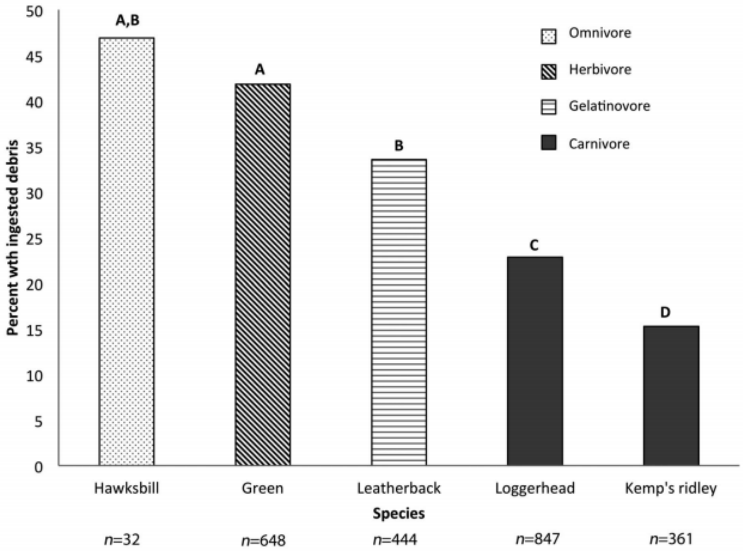

Fig. 1.7.4. Marine debris ingestion rate for each sea turtle species (adopted from Schuyler *et al.*, 2013)
(Note: Flatback turtle was excluded from analyses as it was reported only once.)

Different sea turtles have different life histories and feeding preferences, which influences the probability of marine debris encounter and ingestion by different species (Schuyler *et al.*, 2013). The least understood stage of sea turtle life is post-hatchling and early juvenile stages, and for some species the location and duration of these are unknown (Bolten, 2003). Carr (1987) referred to it as ‘the lost years’.

Bolten (2003) described three different life history patterns regarding the juvenile stage:

- **Neritic developmental pattern -** complete development in inshore marine environment up to depths of 200m (neritic zone); e.g. flatback turtle;
- **Oceanic-neritic developmental pattern -** early juvenile development in the open ocean (oceanic zone) and later in the inshore waters; e.g. loggerhead turtle, potentially hawksbill and Kemp’s ridley turtles;
- **Oceanic developmental pattern -** complete development in the open ocean; e.g. leatherback and olive ridley turtles.

For example, loggerhead turtles, after hatching, leave the beach and enter coastal waters. They move from the neritic to the oceanic zone, where they spend the next 7-11.5 years. After this period, the turtles return to the neritic zone to complete their development (Bolten, 2003). During the oceanic stage, the turtles are exposed to the influence of ocean currents and are drawn into the convergence zones, together with plastic debris and lost fishing gear. This increases the possibility of marine debris ingestion in the oceanic zone (Carr, 1987). Schuyler *et al.* (2012) stated that smaller oceanic-stage turtles feeding on pelagic organisms are more likely to ingest debris than the larger benthic feeders from the neritic zone.

Sea turtles’ diet is quite diverse and it includes algae (adult green turtles), crustaceans and mollusks (loggerhead turtle), soft-bodied invertebrates (flatback turtle), and sponges (hawksbill). Leatherback turtles feed entirely on jellyfish and other gelatinous organisms, while hawksbill feeds on sponges and algae (Schuyler *et al.*, 2013). Schuyler *et al.* (2014) explored the reasons of plastic ingestion by sea turtles offering two hypotheses: i) opportunistic feeding and ii) mistaken identity. Turtles can see colours and they are primarily visual predators, which was also supported by the observational study of feeding behavior of loggerhead turtles by Narazaki *et al.* (2013). Schuyler *et al.* (2014) showed that sea turtles tend to ingest items which are less bright, more flexible and more translucent than the objects in the surrounding environment. They also do not tend to eat blue items. However, the selectivity of plastics by turtles was shown to be most strongly related to flexibility and translucency of the plastic materials, which supports the hypothesis of mistaken identity – 'jellyfish hypothesis'. The turtles choose plastic because it resembles their natural prey, the jellyfish. However, there are reports which indicate that some sea turtles are also indiscriminate feeders, as they sometimes ingest, for example, large amounts of fishing line (Carr, 1987). The items most commonly ingested by sea turtles are shown in the Fig. 1.7.5. Debris size can vary from small plastic pieces (0.5-3cm) to large pieces of bags (>15cm) (Carman *et al.*, 2014).

Sea turtles commonly ingest wrappers, hard plastic pieces, plastic bags and balloons (Carman *et al.*, 2014). In the Northern Australia, weather balloons are often found on the beaches and in the ocean. Whiting (1998) suggested that these balloons should be made of biodegradable materials because they pose a threat to marine life since they resemble jellyfish when floating in the ocean. O'Shea *et al.* (2014) collected 2,460 pieces of these balloons in the Great Barrier Reef World Heritage Area. They modeled the dispersal of weather balloons from 30 field weather stations in Australia situated within 50 km of the coast. If an average of 2.3 balloons is released per day, potentially as much as 180 $m^2$ of latex could end up in the environment each day. If only 50% of these balloons reach the ocean, an estimated 32,850m2 of latex could enter Australia's marine environments from this source annually.

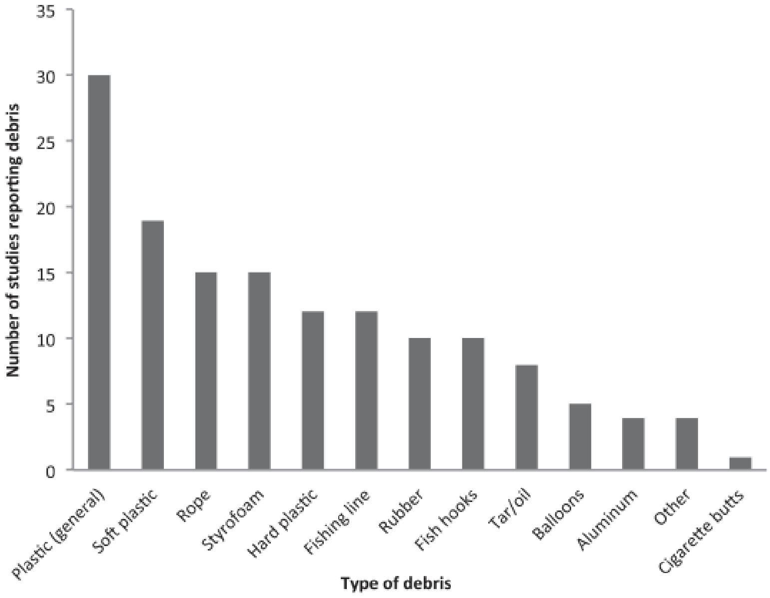

Fig. 1.7.5. Number of studies reporting different types of ingested debris (adopted from Schulyer *et al.*, 2013)

The lethal and sublethal effects of plastic ingestion on individual turtles are similar to the effects debris has on seabirds, and include blockage, ulceration, starvation, rupture and the adverse effects of toxins related to plastic debris (Gregory, 2009). Lutz (1990) carried out an experiment of feeding sea turtles with balloons. He noticed that the blood glucose levels were lower for nine days after ingestion, implying that latex interferes with energy metabolism or gastrointestinal function. In most studies, the majority of ingested debris was found in the intestines, less in the stomach and very little in the oesophagus (Tourinho *et al.*, 2010; Stahelin *et al.*, 2012; Camedda *et al.*, 2013; Campani *et al.*, 2013; Carman *et al.*, 2014). Stahelin *et al.* (2012) reported an extreme case of debris ingestion by a juvenile green turtle, stranded in Florianopolis, Brazil in 2010. The turtle contained 3,593 plastic items weighing 0.3kg. The authors proposed debris ingestion as a likely cause of death due to obstruction of digestive tract and consequent starvation.

The studies on marine debris ingestion by sea turtles in the Pacific region are summarised in the Table 1.7.2. Schuyler *et al.* (2013) analysed studies based on data from 1885 to 2012 (published from 1985 to 2012) and reported that plastic ingestion by sea turtles has different trends for different species. Significant increase was found for green and leatherback turtles and a non-significant increase for loggerhead turtles. Ingestion by Kemp's ridley turtles did not show changes over time, while hawsbill turtles exhibited decrease in plastic ingestion from 1985 to 2012. However, there were only 2 studies on debris ingestion by hawksbill turtles, thus potentially the results might be the artefact of a small sample size. Mrosovsky *et al.* (2009) reported that, in the period between 1885 and 2007, 34% of examined leatherback turtles exhibited plastic ingestion, with the first record of ingestion in1968. Schuyler *et al.* (2013) did not find patterns in debris ingestion geographically nor relation between areas of high debris concentrations and greater incidences of debris ingestion. Sea turtles are migratory species and ingested debris may retain in the gastrointestinal tract for longer periods of time due to inability to regurgitate and expulse, as well as the slow debris decomposition, thus the area of stranding does not necessarily associate with the area of debris ingestion (Schuyler *et al.*, 2013).

### *1.7.1.3. Marine mammals*

Reports of plastic ingestion by marine mammals mainly refer to cetaceans and often are a part of a study on diet of a particular species (e.g. Evans and Hindell, 2004). Early reports date from the 1960s (Gaskin and Cawthorn, 1967). The most usual method for stomach analysis of marine mammals is a necropsy of a stranded animal (e.g. Jacobsen *et al.*, 2010). Stomach flushing has been reported as well (Stamper *et al.*, 2006). Highly controversial lethal method for studying cetaceans has been used by Japanese program JARPA I and II, under the term of 'scientific whaling'. Their annual reports include the results of stomach content analyses and marine debris ingestion was reported in 8 of 25 reports from 1988 to 2013. For example, Cruise report 1990/1991 reports debris ingestion of 1 wooden and 2 plastic pieces by two out of 439 slaughtered minke whales (Ishikawa *et al.*, 2002).

Thirty out of 115 marine mammal species (26%) have the tendency to ingest marine debris (Secretariat of the Convention on Biological Diversity and the Scientific and Technical Advisory Panel – GEF, 2012). A more recent review, by Baulch and Perry (2014), reports that 48 cetacean species (56% of all cetacean species) are prone to marine debris ingestion. In general, most reports on debris ingestion by marine mammals refer to cetaceans, and much less to sirenians and pinnipeds, or other marine mammal groups, such as the sea otters and polar bears. Deep diving toothed whales (Odontoceti), such as the sperm and pygmy sperm whales, and beaked whales, seem to be especially vulnerable to plastic ingestion (Simmonds, 2012). Frequently reported sperm whale (*Physeter macrocephalus*) (Roberts, 2003; Evans and Hindell, 2004; Fernandez *et al.*, 2009; Jacobsen *et al.*, 2010; Mazzariol *et al.*, 2011) is classified as vulnerable according to IUCN Red List. Along with deep divers, dolphins and porpoises have been reported quite often as well (see Table 1.7.3.). Reports on debris ingestion by baleen whales (Mysticeti) is much less common than by toothed whales. Apart from cetaceans, there is very little information about ingestion by sirenians (Beck and Barros, 1991) and pinnipeds (Eriksson and Burton, 2003; Raum-Suryan *et al.*, 2009; Rebolledo *et al.*, 2013).

Marine mammals have been documented to ingest a variety of objects. Walker and Coe (1990) noted that the majority of foreign objects ingested by cetaceans are plastics; comprising 80% of all ingested debris in 43 stranded odontocetes with 62.5% being plastic bags and sheeting, and 17.5% drinking straws, bottle caps, discarded fishing nets, synthetic ropes and plastic containers. De Stephanis *et al.* (2013) reported ingestion of various plastic items by a single sperm whale in Spain coastal area in 2012, such as plastic cover material, burlap plastic bags, flower pots, hosepipes and ropes, which originated from the local greenhouse agriculture. The whale had ingested 59 items weighing 18kg and covering an area of 37.5m$^2$. Poncelet *et al.* 2000 (in Baulch and Perry 2014, p. 3) reported an ingestion incidence of 378 items (33kg) ingested by a Cuvier's beaked whale. Two baleen whales, Bryde's whale stranded in 2000 in Cairns, Australia (Townsville Bulletin, 2001, in Simmonds 2012, p. 5), and minke whale stranded in 2002 in Normandy France (Gerard, 2002), ingested large number of plastic bags, 30 and 16, respectively.

Fishing-related debris is also commonly ingested by marine mammals. Two sperm whales stranded in 2008 on the Californian coast contained astounding amounts of fishing nets in their stomachs, 59 and 105 pieces in each, which weighed 22.2kg and 57.1kg, respectively (Fig. 1.7.6.) (Jacobsen *et al.*, 2010). Levy *et al.* (2009) reported ingestion of nylon filaments and nets by a common bottlenose dolphin in 2002 in the Port of Haifa, Israel. Beck and Barros (1991) studied and collected dead manatees from 1978 to 1985 and found that 63 of 439 animals ingested debris, most commonly monofilament fishing line.

Fig. 1.7.6. Ingestion of 105 pieces of net by a sperm whale stranded in March 2008 in California (case study described in Jacobsen *et al.* (2011)), photo credit: Chris Whittier, Marine Mammal Center.

Secondary ingestion of plastics by fur seals was proposed by Eriksson and Burton (2003). They examined fur seal feces in two occasions (1990/91 and 1996/97) in Macquarie Island and found 164 pieces of plastic in 145 fur seal scats. The plastic items were mainly fragments, which were likely consumed by pelagic fish *Electrona subaspera*, thereafter eaten by fur seals.

The causality of marine debris ingestion by marine mammals has not been extensively discussed. Walker and Coe (1990) suggested that odontocetes, probably being discriminating feeders in the wild, most likely ingest marine debris incidentally while feeding on benthic prey (e.g. sperm whale and Baird's beaked whale. They also stated that it is unlikely that debris is ingested due to mistaken identity, as odontocetes use echo-location for feeding. Simmonds (2012) noted that these deep diving odontocetes, which use a suction feeding technique, are of particular susceptibility to debris ingestion. Furthermore, di Beneditto and Ramos (In press) found that Franciscana dolphin, which is a near-bottom feeding species, exhibited ingestion in 14.4% of 89 examined specimens, while ingestion by Guiana dolphin, a near-surface feeder, occurred in 1.3% of animals. A grey whale, stranded in 2010 in Seattle, US, contained large amounts of debris, and is a bottom-feeding baleen whale using a suction technique to extract the invertebrates from the sediment (Cascadia Research, 2014). Other reported baleen whales, which are mainly surface filerfeeders, also most likely ingest debris by accident. However, the arguments for incidental ingestion do not explain the cause of debris ingestion by surface selective feeders, such as dolphins and porpoises.

Ingested macro-plastics may leave no apparent pathological effects, but it was regularly documented that it may cause complete obstruction of the gastrointestinal tract and internal injuries, which leads to malnutrition, starvation, diseases associated to reduced immunity, reduced reproductive success and quality of life and, finally, mortality (Baulch and Perry, 2014). Fossi *et al.* (2012) studied ingestion of microplastics by Mediterranean fin whales (*Balaenoptera physalus*) by determining the levels of phthalates (plasticising additives in plastic) in the blubber of stranded animals as an indicator of microplastic ingestion. Phthalates are potentially endocrine disrupting chemicals as well.

With respect to cetacean mortality due to debris ingestion, Baulch and Perry (2014) reported that seven of 21 species succumb to ingestion in between 1.3% and 22.2% of cases. The greatest mortality rates were recorded for Gervais beaked whale (16.7% of 12 and 22.2% of 9 animals), while for rest of the seven species the rates ranged between 3 and 10%. The study of Beck and Barros (1991) on Florida manatees showed that marine debris caused death of 4 of 439 manatees in a span of 8 years (mortality rate of 0.9%).

Cetacean gastrointestinal tracts are different to other marine mammals. Their stomach comprises four compartments (similar to cattle, goats and sheep) called the fore-, main, connecting and pyloric stomachs (Berta *et al.*, 2006; Mead, 2008). The forestomach is present in all mysticetes and most odontocetes, except ziphiides and franciscana. It holds glands and is lined by keratinized epithelium. In mysticetes, the forestomach often contains pebbles for grinding fish bones and crustaceans (Berta *et al.*, 2006). The main and connecting stomachs are present in all cetaceans. The former (also fundic chamber) is enveloped by folded mucosa with gastric glands. It produces most of the digestive enzymes and acids. The pyloric stomach is lined by smooth mucosa with pyloric glands. Digestion is still active in the last stomach, duodenum and small intestines.

Walker and Coe (1990) provided necropsy details of a captive dolphin, which died due to ingestion of a plastic bag. It remained mainly in the forestomach, but extended to sphincter and main stomach as well, causing inflammation of the tissue. They reported that almost half of the forestomach lining and submucosa was eroded with the underlying necrotic tissue protruding deep in the stomach muscles. Edema was evident in the stomach wall and serosa, which were 5-6 times thicker than normal. They suggested that the plastic bag caused redundant excretion of gastric fluids, damaging the stomach lining. For example, Fig. 1.7.7. shows an illustration of a piece of plastic inside the stomach chambers of a pygmy sperm whale (*Kogia breviceps*) stranded in 1993 in New Jersey (Stamper *et al.*, 2006).

Stamper *et al.* (2006) described a rare example of a complete recovery of a whale from a severe plastic ingestion and multiple parasitic infections. The animal was a juvenile female pygmy sperm whale found stranded in emaciated state in 1993 in New Jersey. Parasitological examination showed several different species of nematode and trematode ova in feces and bacteria, flagellates, ciliates and yeast in the blowhole. Moreover, after gastroscopy identified the presence of foreign objects, the stomach was flushed several times until all of the plastic was removed. It included plastic bags, cellophane and cigarette box wrappers. The animal fully recovered after six months and was tagged and released back to the sea.

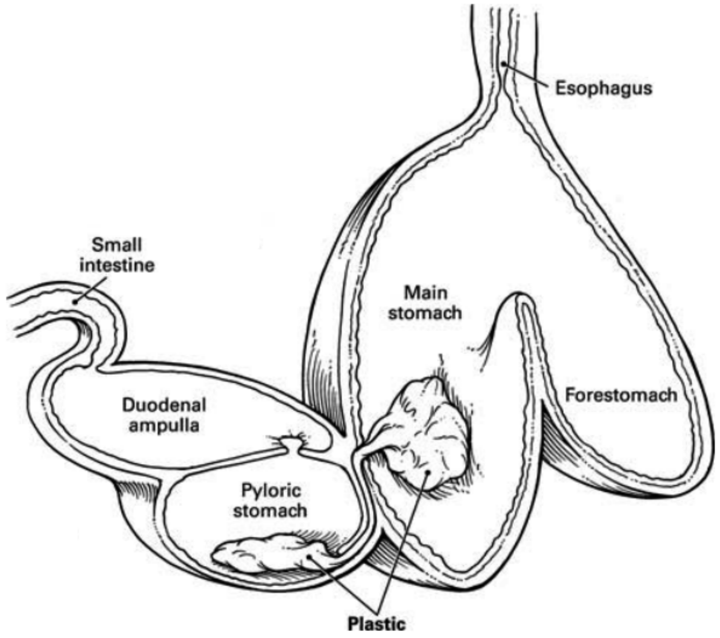

Fig. 1.7.7. Plastic in the stomach of a pygmy sperm whale (Stamper *et al.*, 2006).

Walker and Coe (1990) reviewed the cases of plastic ingestion by odontocetes. They provided detail information on ingestion by 43 animals, stranded between 1963 and 1986 on the west (58%) and east (37%) coast of North America, one case from the Gulf of Mexico and one from Hawaii. Most frequently reported odontocetes were dolphins and porpoises, the great majority of which were stranded on the Californian coast. One of nine rough-toothed dolphins (*Steno bredanensis*) stranded in Maui, Hawaii, 1976, had a plastic bag in its stomach (Table 1.7.3).

Baulch and Perry (2014) gathered and analysed previous reviews and records of marine debris ingestion and entanglement incidences by cetaceans. Marine debris ingestion was reported to occur with least 462 individual cetaceans belonging to 48 species – nine mystecetes and 39 odontocetes. There has been an increase in the number of debris ingestion incidences over time, occurring 11 times more often than in the 1960s (Fig. 1.7.8.).

Table 1.7.4. provides details of reports and studies on debris ingestion from the Pacific area. In accordance with the prevalence in other studies worldwide, sperm whale was most commonly reported species in the Pacific area as well.

Baulch and Perry (2014) pointed out that the sample size is important in determination of the occurrence of debris ingestion for a particular species. A small sample size may give either an underestimate or overestimate of ingestion rates. For example, 1 of 3 stranded sei whales on the UK coast contained marine debris, resulting in a 33% ingestion rate. The authors suggest that a large sample size should be used whenever possible, which is, understandably, not feasible for rare species.

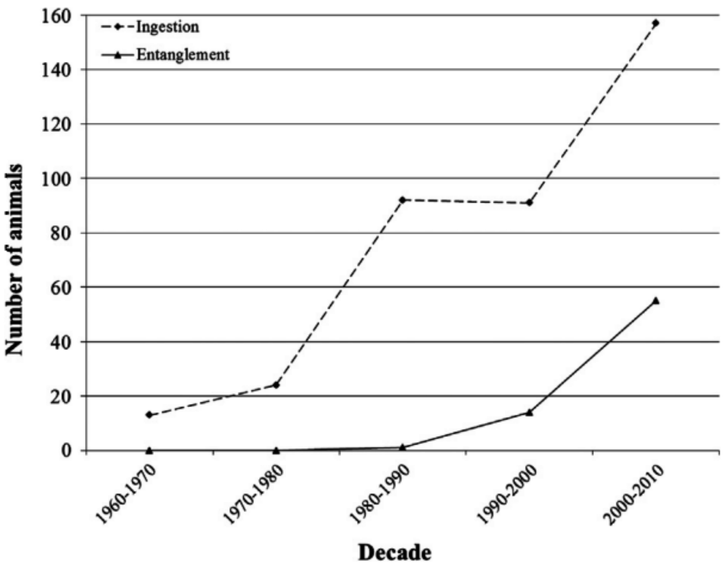

Fig. 1.7.8. Number of documented debris interactions involving cetaceans from 1960 to 2010 (adopted from Baulch and Perry 2014).

### *1.7.1.4. Fish*

Early documentation of plastic ingestion by fish indicated a quite high incidence of occurrence (Table 1.7.5.). Carpenter *et al.* (1972) investigated the pollution of surface waters of the northwest Atlantic by plastic polystyrene spherules and ingestion by fish. They examined 14 species and found that 8 ingested plastic, of which the greatest ingestion rate was observed in white perch and silversides (33%). They also reported ingestion by winter flounder and grubby larvae, only 5mm in size, which had ingested 0.5mm large spherules. Anon (1975) found plastic cups in 4 species, namely pollack, coalfish, cod and pout whiting, in the English Channel. One pollack contained 4 plastic cups. Kartar *et al.* (1973, 1976) reported the occurrence of polystyrene spherules in the intestines of 4 fish species (flounders, sand goby, sea-snail and five-bearded rockling). Some young flounders (2-5cm) ingested as much as 30 spherules.

Secretariat of the Convention on Biological Diversity and the Scientific and Technical Advisory Panel – GEF (2012) reported that 41 fish species out of 16,754 known fish species ingest marine debris. However, after reviewing more recent papers this number increased for another 24 species. Table 1.7.5. comprises 65 fish species, 50 osteichthyes (Actinopterygii) and 15 elasmobranchs (Elasmobranchii), reported for plastic ingestion in the wild, while laboratory experiments are not included (e.g. Drenner *et al.*, 1984; Mummert and Drenner 1986; Rochman *et al.*, 2013). Osteichthyes exhibit greater debris ingestion rates than elasmobranchs, which are generally of higher risk conservation status by IUCN Red List than osteichthyes. Anastasopoulou *et al.* (2013a) found that 5 out of 26 examined species from the Eastern Ioanian Sea ingested debris. They examined 1502 individuals and 28 contained debris (1.9%). Ingestion incidence was much greater in elasmobranchs than osteichtyes, 5 out of 9 elasmobranchs and only 1 out of 17 osteichtyes species. The authors found 1.3 pieces of debris on average per examined specimens.

Cliff *et al.* (2002) analysed stomach content of 15,666 large sharks caught in the protective nets on the KwaZulu-Natal coast between 1978 and 2000. Sixty sharks from 10 species ingested debris. The greatest ingestion rate occurred with tiger sharks, listed as near threatened by IUCN. Although the ingestion rate by other 9 species was quite low (<1%), 5 species are considered vulnerable and 1 endangered (See Table 1.7.5.).

Plastic ingestion has been studied on fish of different habitat and feeding preferences, including deep water fish and and demersal fish (Anastasopoulou *et al.*, 2013; Foekema *et al.*, 2013; Lusher *et al.*, 2013), estuary fish (Possatto *et al.*, 2011; Dantas *et al.*, 2012; Ramos *et al.*, 2012), epipelagic and mesopelagic (Boerger *et al.*, 2010; Davison and Asch, 2011; Choy and Drazen, 2013; Jantz *et al.*, 2013; Lusher *et al.*, 2013). A study on difference in debris ingestion by epipelagic and mesopelagic fish showed that, surprisingly, mesopelagic fish ingested more debris than the epipelagic fish (Choy and Drazen, 2013). The authors provided several possible explanations. Firstly, assuming that the plastics mainly float at the surface of the ocean, previous diet studies potentially overlooked the feeding pattern of these supposedly mesopelagic fish, which in fact may also feed at surface at night. Secondly, mesopelagic fish may feed on prey which ingested debris, however the authors overruled this possibility as the ingested plastic was much larger (>~4cm) than the plastic found in prey species (<3mm). Thirdly, fish may have ingested plastic while being hauled on the boat, but if this was the case, other fish would demonstrate higher incidence of ingestion as well. Lastly, the ingestion of plastic occurred at depth, since not all plastic is positively buoyant or it can be in the process of sinking due to biofouling or wind-mixing of water layers. Lusher *et al.* (2013) found no significant difference in ingestion rates of pelagic and demersal fish. Generally, more detailed analysis of debris ingestion by fish with different feeding preferences, habitat and ontogenic phases is needed to draw more reliable conclusions on which species might be more vulnerable to plastic ingestion.

Fish commonly ingest unidentifiable plastic fragments, generally smaller in size, often <5mm (Boerger *et al.*, 2010; Davison and Asch, 2011; Dantas *et al.*, 2012; Foekema *et al.*, 2013; Lusher *et al.*, 2013), but sometimes, depending on the size of the fish, larger items are ingested as well (e.g. Cliff *et al.*, 2002; Haetrakul *et al.*, 2007; Anastasopoulou *et al.*, 2013; Choy and Drazen, 2013). Anastasopoulou reported 86.5% of ingested debris was plastic. Several studies investigated the ingestion of plastic spherules (Carpenter *et al.*, 1972; Kartar *et al.*, 1976). Cliff *et al.* (2002) found that the majority of plastic objects ingested by sharks off South Africa were plastic bags (48%), plastic sheeting (18%) and plastic bottles and containers (11%). Estuarine fish were reported to ingest polyfilament nylon fragments (Dantas *et al.*, 2012; Ramos *et al.*, 2012).

Carson (2013) examined 5,518 plastic items, weighing 67.5kg, collected from Kamilo Point, Hawai'i in 2011 and 2012, for the presence of obvious bite marks (Fig 1.7.9.). Apparent attack signs were present at 15.6% of all items (871) or 17.3% by weight (11.7kg). More than 14% of large plastic items were obviously bitten before stranding on the beach. If extrapolated to one year, 17.3% of all plastic debris collected in that area annually, would represent 1.3 tonnes of plastic debris attacked by marine animals in one year. Yellow, blue and green objects had bite marks more often than black, grey, while and clear. The tooth mark sizes ranged from <1 to 20.5mm, most commonly being 2-4mm. The authors did not relate the bite marks to a specific species or life stage.

Carson (2013) distinguished 2 types of bite marks, large bite marks or missing portions, and small or 'nibbling' bite marks. The large bite marks are likely made by large predators which mistaken the object for its prey and the small bite marks might belong to fish which feed on organisms fouling the floating object. Ramos *et al.* (2012) studied ingestion of nylon fragments by estuary fish in Brazil and they offered 3 possible causes of ingestion: i) secondary ingestion, ii) incidental ingestion along with the sucked sediment and iii) incidental ingestion while feeding on organisms growing on nylon fragments.

Presumably the effects of plastic ingestion are similar to ingestion by other vertebrates. Haetrakul *et al.* (2007) reported ingestion of a single plastic straw by a whale shark in 2005 in Thailand as a potential cause of death, due to multiple lacerations of the stomach wall reaching into the sub-serosal stomach layers, coupled with marked hemorrhaging. Ramos *et al.* (2012) observed that fish which contained plastic in the stomach also had less gut content (i.e. food). Okada *et al.* (2014) described high mortality in juveniles bluefin tuna farmed for aquaculture due to ingestion of inorganic material, such as styrofoam and wood. Mortality rates after 28 days were 45.7% and almost half of these mortalities occurred within the first 3 days.

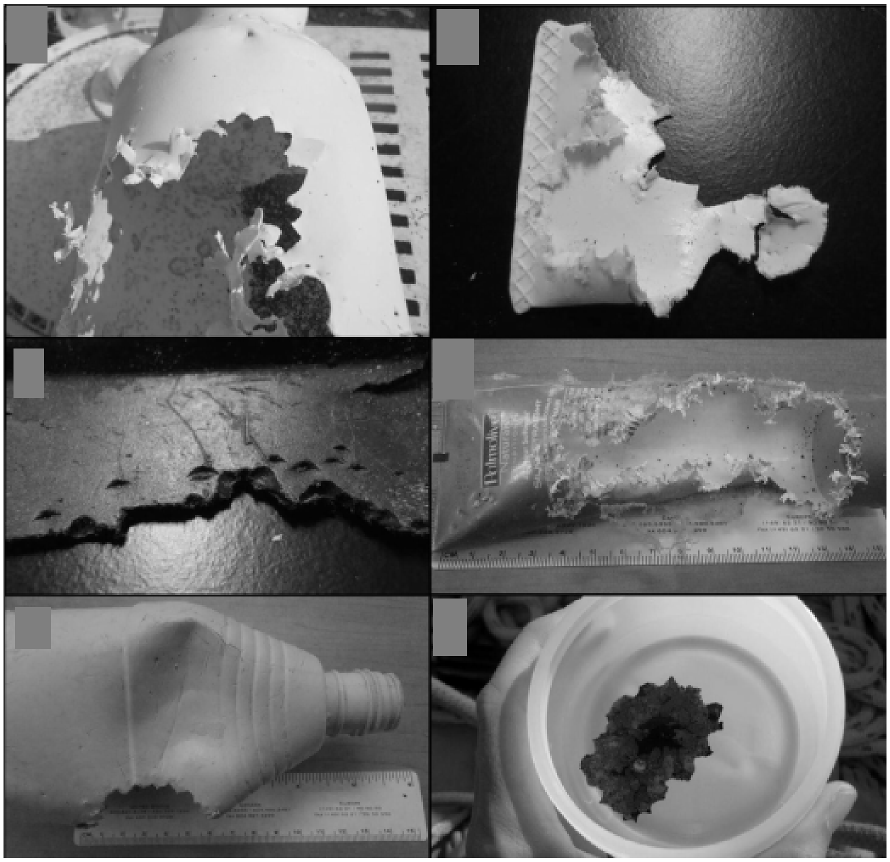

Fig 1.7.9. Examples of bite marks on different plastic items collected from Kamilo Point, Hawaii (in Carson (2013)).

### *1.7.1.5. Invertebrates*

Holmstrom (1975) identified 'eating traces' on plastic film from the bottom of Skagerack, Sweden, and proposed that the traces were probably made by a herbivorous mollusc, which ingested plastic incidentally while scraping of the biofilm of brown alga *Lithoderma sp.* growing on the plastic film. Carpenter *et al.* (1972), as a part of their study on ingestion of plastic spherules by various fish species, also found one small spherule (0.6mm) in a 20mm-long arrow worm *Parasagita elegans*.

Subsequently, plastic ingestion was found to occur in numerous invertebrates (Table 1.7.6.). Studies on ingestion mainly focused on laboratory feeding experiments, rather than ingestion occurring in the wild. Cole *et al.* (2013) studied ingestion of polystyrene microbeads (0.4-30.6 µm) by various zooplankton of 26 species in different life stages. Half of the species (13) were found to ingest plastic (Fig. 1.7.10.).

Studies on ingestion in the wild yielded quite interesting results. Mathalon and Hill (2014) compared ingestion rates by wild mussels collected from Eastern passage of Nova Scotia, Halifax, with the mussels from aquaculture site, 800km away from Halifax, off the west coast of Newfoundland and Labrador. Farmed mussels exhibited greater ingestion rates than wild mussels (75 and 34 pieces of plastic per individual, respectively). Murray and Cowie (2011) examined ingestion rates in the wild and in controlled conditions. They found that 100 in 120 (83%) commercially harvested Norway lobsters taken from Clyde Sea, Scotland, ingested plastic, while 100% of the animals of the same species ingested plastic during experimental feeding.

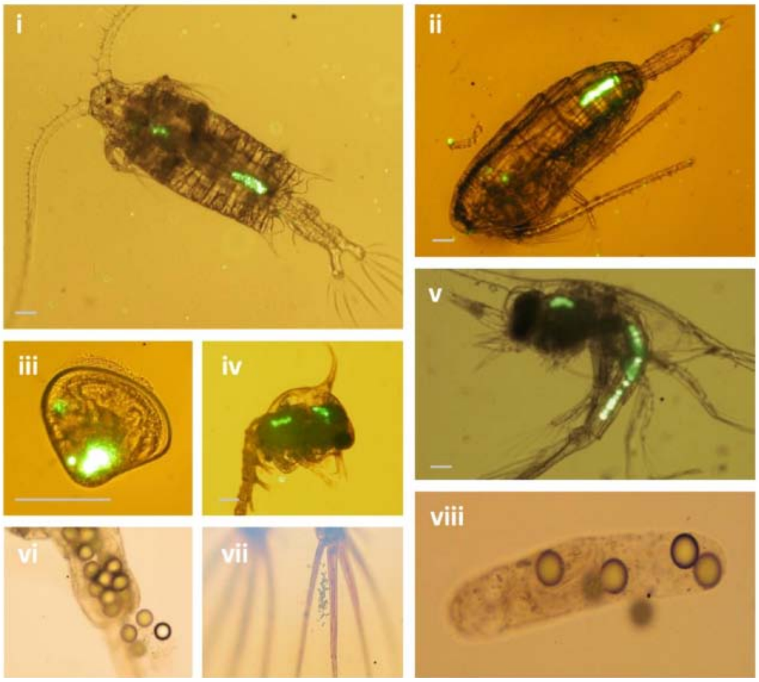

Fig. 1.7.10. Ingestion of fluorescent plastic beads by various invertebrates: (i) the copepod *Centropages typicus* - dorsal view (7.3 µm polystyrene (PS) beads); (ii) the copepod *Calanus helgolandicus* - lateral view (20.6 µm PS beads); (iii) a D-stage bivalve larvae - dorsal view (7.3 µm PS beads); (iv) a Brachyuran (decapod) larvae (zoea stage) - lateral view (20.6 µm PS beads); (v) a Porcellanid (decapod) larvae - lateral view, (30.6 µm PS beads); (vi) 30.6 µm PS beads in the posterior-gut of the copepod *Temora longicornis* during egestion, (vii) 1.4 µm PS beads trapped between thefilamental hairs of the furca of *C. typicus*; (viii) a *T. longicornis* faecal pellet containing 30.6 µm PS beads

Goldstein and Goodwin (2013) investigated plastic ingestion by barnacles attached to floating marine debris in the North Pacific Subtropical Gyre. The study was carried out as a part of SEAPLEX expedition (Scripps Environmental accumulation of Plastic Expedition) from Honolulu to San Francisco. The authors examined gastrointestinal tracts of 385 barnacles, of which 129 contained plastic (33.5%). A total of 518 particles were obtained from 129 barnacles (av. 4p/individual), with a maximum of ingested plastic particles per individual found being 30. Blockage or accumulation of particles in the digestive system of the animals was not observed and the particles were small enough to be successfully egested.

Several experimental studies did not find noticeably detrimental effects of ingested plastic on tested animals (e.g. Kaposi *et al.*, 2013; Ugolini *et al.*, 2013). Some invertebrates, such as adult amphipods (Ugolini *et al.*, 2013) or sea urchin larvae (Appelmans, 1994; Kaposi *et al.*, 2013) tend to eject or egest plastic particles. For example, amphipods expel most of the ingested microspheres within the first 24 hours and the rest within a week.

However, a number of studies reported adverse changes in affected individuals. Von Moos *et al.* (2012) used HDPE particles (0-80µm) to investigate microsplastic uptake by mussels. They detected strong inflammatory response in the form of increase in the formation of granulocytoma and decrease in lysosomal stability. The authors also found microparticles inside epithelial cells of the primary and secondary duct and in tubules of the digestive gland. They proposed that the particles entered through the mouth opening, were carried to gastrointestinal tract where they were internalised into the cells of the digestive system by endocytosis.

Chan and Witting (2012) observed size-specific ingestion by salps, where only salps larger than 3cm ingested the plastic particles, whereas the smaller salps concentrated particles in the oral opening, potentially clogging up the feeding apparatus. Wright *et al.* (2013) noted that plastic significantly decreased energy reserves in lugworms (infauna) by up to 50%, while Besseling *et al.* (2012) showed that ingestion of polystyrene microplastic caused reduction in feeding activity in lugworms as well as bioaccumulation of POPs. Farrell and Nelson (2013) studied the trophic transfer of microplastic from mussels to crabs and found microplastic in haemolymph of the crabs. The highest concentration of 15,033 microspheres per mL was at 24h, and decreased to 267 p mL$^{-1}$ after 21 days.

### *1.7.2. Entanglement and ghost fishing*

According to the report by Secretariat of the Convention on Biological Diversity and the Scientific and Technical Advisory Panel – GEF (2012), entanglement in marine debris is 34% more frequently reported than ingestion, as well as direct harm or death by entanglement, in 80% of reports, as opposed to 5% caused by ingestion. The authors, however, indicated that this might be due to a bias in reporting the incidences, as entanglement cases are easily visible externally, while debris ingestion can only be detected by necropsy. Furthermore, Laist (1997) emphasised that generally the data on entanglement are ‘opportunistic anecdotal records’ and that any attempts of systematic sampling usually resulted in sample sizes too small for appropriate statistical analyses.

The deleterious effects of entanglement in fishing or other marine debris include strangulation, drowning, infection or necrosis of the tissue damaged by entangling objects (e.g. loops, ropes, fishing lines, nets), reduced swimming efficiency due to injuries and/or drag, limited foraging ability and starvation (Moore, 2008, Baulch and Perry, 2014). Impacts of entanglement on population levels have been rarely reported (e.g. Fowler and Baker, 1990), most likely because, in order to infer on these effects, long-term studies, which have not been completed, are required.

Marine animals are particularly susceptible to the impacts of fishing debris (Matsuoka *et al.*, 2005). Around the time the fishing industry completed the shift from the use of natural to synthetic fibres, early evidence of seabird, fish and seal entanglement in this improved synthetic fishing gear, when lost in the marine environment, was documented (Degange and Newby, 1980; Shaugnessy, 1980).

Most entanglement reports include references to lost or discarded fishing gear. As one example, Kim *et al.* (2014) provided a national annual estimate of abandoned traps and gillnets in South Korea of 11,436 and 38,535 tonnes, respectively. Often, various fishing nets, monofilament line, traps and pots can continue fishing after being discarded, lost or abandoned at sea, which occurs due to adverse weather conditions, propeller strikes, snags on obstacles, poor maintenance or vandalism (Uhlmann and Broadhurst, 2013). This phenomenon is known as ‘ghost fishing’ and is defined as ‘the ability of fishing gear to continue fishing after all control of that gear is lost by the fisherman’ (Smolowitz, 1978, p. 3). The problem was first identified and discussed at a Food and Agriculture Organization (FAO) meeting in Rome in 1960 (Smolowitz, 1978). Matsuoka *et al.* (2005) noted that the above definition of ghost fishing should include the mortality of marine organisms caused by abandoned fishing gear. For example, the authors conducted underwater surveys which identified various derelict fishing gear, including gillnets, trammel-nets, small-scale Danish seines, bottom longlines, cage traps and octopus pots, however mortality was attributed only to some of the gear, not including octopus pots and bottom longlines.

The magnitude of the risk associated with derelict fishing gear has been assessed through experiments, as well as direct measurements and observations of the actual catch in derelict nets and traps. Gerrodette *et al.* (1990) and Matsumura *et al.* (1990) studied the movement and fate of experimentally deployed drift fishing nets in the North Pacific. The nets varied in sizes: 50m, 100m, 350m, 1000m (Gerrodette *et al.*, 1990) and 2000m (Matsumura *et al.*, 1990). In both studies the nets were tracked by radio-satellite buoys using close-range radio tracking and satellite tracking by Argos system. Their studies showed that floating nets changed their shape considerably in the first couple of days after releasing. Matsumura *et al.* (1990) found that initially 2000m long nets change their shape within the first 10 days of floating and bundle up into a 50m clump. Gerrodette *et al.* (1990) also reported that the nets folded into smaller accordion-like shapes; however the nets were still hanging 9m in depth. Both studies reported similar floating speed of nets, between 10-20km day$^{-1}$ (Matsumura *et al.*, 1990) or around 15km day$^{-1}$ (Gerrodette *et al.*, 1990). Gerrodette *et al.* (1990) noted that very little marine life was trapped by the nets in the first 10 days of close observation, a small marlin and a large flyingfish, while several seabirds and fish were observed investigating the nets.

Several authors estimated the abundance and composition of the catch of experimental nets in different conditions. Ayaz *et al.* (2006) investigated the difference in fishing ability of lost monofilament and multifilament gillnets. Six nets of each type were deployed at 2 locations in Izmir Bay, Turkey, in 2002, at a depth of 9-14m. In just over 100 days (monofilament 106 days and multifilament gillnets 112 days), the nets caught almost 400 specimens of 29 different species: 22 fish species, 5 crustacean, 1 cephalopod and 1 gastropod species. Most commonly trapped species include sea bream (*Diplodus annularis*), painted comber (*Serranus scriba*), red scorpion-fish (*Scorpaena scrofa*) and spider crab (*Maja* spp.). It should be noted that the set of nets from one location was lost after 42 days, thus the comparison between locations was not possible. Monofilament nets caught significantly more catch than multifilament, however both nets demonstrated exponential decline in fishing ability due to biofouling and deposition of detritus. After 6 weeks, the nets lost their ability to fish by more than 50%. A year after the deployment, multifilament nets completely collapsed and monofilament was heavily biofouled and was about to collapse.

Kaiser *et al.* (1996) found the change in the composition of catch over time. They used one gillnet and one trammel net 1000m offshore in St. Bride’s Bay, UK, which both collapsed the first day of installation due to a large quantity of dogfish (*Scyliorhinus canicula*) trapped in the nets. The decomposing fish attracted scavenging crustaceans (e.g. *Maja squinado* and *Cancer pagurus*), which finally also became entangled in the collapsed nets. This ‘self-baiting’ of the nets continued to attract organisms, however these nets exhibited decline in fishing ability over time as well, due to decrease in fishing area of the nets. The authors also observed many starfish (*Echinaster (Echinaster) sepositus* and *Marthasterias glacialis*) and sea urchins (*Echinus esculentus*) subsequently feeding on decaying remains in the nets. The dominant species entangled in the nets were mainly elasmobranchs (*Scyliorhinus canicula* and *Scyliorhinus stellaris)* and crustaceans (*Maja squinado*).

Baeta *et al.* (2009) compared ghost fishing capacity of trammel nets on different substrata, sandy and rocky bottoms, off the Portuguese central coast at depths between 15 and 25m. The nets fished for 285 days. Their fishing area decreased about 40% in the first 30 days, followed by a sharp decrease on the sandy and gradual on the rocky bottoms. More than 33 species were captured in the nets (28 fish, 3 crustaceans, 2 cephalopods and 1 gastropod), 41 individual per 100m of nets on sandy and 61 individual on rocky bottom.

While in shallow-water experiments, biofouling and sedimentation showed to be important factors in reducing the fishing ability of derelict nets, in the deeper waters these factors are not as relevant. Humborstad *et al.* (2003) completed an experiment on derelict gillnets at depths of 537-851m. Their results indicated that, although the catch generally decreased over time, after 45th day the fishing capacity of the nets stabilised at low catches with the absence of biofouling or damage to the nets. The authors proposed that the only factor contributing to reduction of ghost fishing efficiency at greater depths is the weight of entangled organisms which decreases the fishing area by lowering the headline height.

However, after the clearance of decomposing bodies, the headline might rerise, recovering the lost fishing area of the net.

A case study on 870 derelict gillnets, with residence time of several years in Washington's inland waters, resulted in a large number of entanglement incidences (Good *et al.*, 2010). Almost 34,000 individuals were trapped in gillnets including: 31,278 invertebrates of 76 species, 1,036 fish of 22 species, 514 seabirds of 16 species and 23 mammals of 4 species. The mortality was documented in 56% of invertebrates, 93% of fish and 100% of seabirds and marine mammals. Invertebrate species with high mortality rates were: giant barnacle (*Balanus nubilus*), Dungeness crab (*Metacarcinus magister*), red rock crab (*Cancer productus*), butter clam (*Saxidomus gigantea*), smooth pink scallop (*Chlamys rubida*), Pacific littleneck clam (*Leukoma staminea*) and Nuttall's cockle (*Clinocardium nuttalli*). Fish species which exhibited higher mortality were: salmonid (*Onchorynchus* spp.), spotted ratfish (*Hydrolagus collie*), spiny dogfish shark (*Squalus acanthias*), black rockfish (*Sebastes melanops*), lingcod (*Ophiodon elongatus*) and kelp greenling (*Hexagrammos decagrammus*); of which spiny dogfish shark is listed as vulnerable by IUCN Red List. All seabirds succumbed entanglement and include: cormorants (*Phalacrocorax* spp.), scoters (*Melanitta* spp.), loons (*Gavia* spp.) and grebes (*Podiceps* spp.). Trapped marine mammals also exhibited 100% mortality rates and belong to one of the following species: harbour seal (*Phoca vitulina*), California sea lion (*Zalophus californianus*), harbour porpoise (*Phocoena pocoena*) and river otter (*Lontra Canadensis*).

Uhlman and Broadhurst (2013) noted that traps are potentially more hazardous type of derelict fishing gear than gillnets due to their ability to stay functional for longer periods. Matsuoka *et al.* (2005) noted that, based on long-term observations, traps can sometimes ghost fish for more than 3 years, and in deep-water conditions, with decreased biofouling and physical damage by hydrodynamic activity, even longer.

Al-Masroori *et al.* (2004) measured ghost fishing by experimental fish traps at traditional fishing grounds near Muscat, Oman, at depths between 16 and 36m. Overall catch in 21 weeks, in 25 experimental traps, weighed 434.3kg and included 426 fish and 60 cuttlefish, of which 83% was commercial catch and the remainder by-catch. Estimated daily catch was 1.34kg with a decreasing trend over time. Anderson and Alford (2014) collected and removed 3607 derelict blue crab traps in Louisianna in 2012 and 2013, with the help of volunteers and citizen scientists. In 1073 traps, volunteers identified 19 species. Citizen scientists released 671 live and 61 dead blue crabs and estimated that 65% of the traps were still actively ghost fishing.

#### *1.7.2.1. Seabirds*

According to the Secretariat of the Convention on Biological Diversity and the Scientific and Technical Advisory Panel – GEF (2012), 67 out of 213 seabird species are susceptible to entanglement in marine debris. Assessment of entanglement is generally obtained through at-sea observations (e.g. Degange and Newby, 1980; Rodriguez *et al.*, 2013) or examination of the nests in seabird colonies (e.g. Hartwig *et al.*, 2007; Votier *et al.*, 2011; Bond *et al.*, 2012; Lavers *et al.*, 2013), and sometimes by incidental finds of entangled animals stranded on the coast or entangled in the vegetation (e.g. Yorio *et al.*, 2014).

Degange and Newby (1980) reported 99 entangled seabirds and more than 200 fish in a 1500m long section of an approximately 3500m long salmon driftnet, which was estimated to be adrift at sea, in the North Pacific south of Aleutian Arc, for at least 30 days. The most prevalent seabird species found entangled were Shearwaters (*Puffinus griseus*, *Puffinus tenuirostris*), Tufted puffins (*Fratercula cirrhata*) and Fulmars (*Fulmarus glacialis*). Rodriguez *et al.* (2013) investigated entanglement in Northern Gannets (*Morus bassanus*) in the western Mediterranean and eastern Atlantic using vessel-based transect counts.

The surveys covered the area of 4189km$^2$. In total, 3672 Gannets were counted and 34 (0.93%) of these animals were entangled in predominantly fishing-related debris - ropes, net fragments, nylon fishing lines.

Seabirds use various materials available in the environment for building nests, including more readily available synthetic materials. Hartwig *et al.* (2007) found the increase in the proportion of Kittiwake (*Rissa tridactyla*) nests containing debris, in northwest Denmark, from 39.3% in 1992 to 57.2% in 2005. The birds mainly used synthetic strings, plastic foil and fishing net fragments. Examination of Northern Gannets' nests in Wales, UK, showed that many seabirds also become victims of entanglement in synthetic materials used in nest building (Votier *et al.*, 2011). On average, around 65 birds entangle annually, usually around legs and feet, and sometimes wings as well. Some of the birds were released by the authors who noted that they would certainly die without their intervention. In most reports, synthetic material used for nest building was related to fishing activities and includes ropes, net remnants and fishing line (Votier *et al.*, 2011; Bond *et al.*, 2012; Yorio *et al.*, 2014). Conversely, Lavers *et al.* (2013) found that the most dominant type of debris encountered in nests of Brown Booby (*Sula leucogaster*) in colonies on Ashmore Reef, Timor Sea, were hard plastic pieces, followed by rope and foam.

### *1.7.2.2. Sea turtles*

There is little documentation on sea turtle entanglement in marine debris, however it has been reported that all known sea turtle species (7) are prone to entanglement in marine debris (Secretariat of the Convention on Biological Diversity and the Scientific and Technical Advisory Panel – GEF, 2012). Duronslet *et al.* (1991) reported quite low entanglement rates of 3.5% (6/171) among stranded turtles in Texas and Louisiana between 1987 and 1989, whereas debris ingestion rates were almost 60%. Conversely, Sadove and Moreale (1990) found greater occurrence of entanglement of sea turtles (58 individuals) than ingestion (14 individuals) in New York Bight between 1979 and 1988.

Casale *et al.* (2010) analysed a total of 5,938 records of stranded or floating loggerhead turtles (*Caretta caretta*) in Italian waters from 1980-2008. They affirmed that entanglement in marine debris (4.6-6.6%) has an impact on these turtles; however the main cause of mortality related to human activities was attributed to interactions with active fisheries. A study on bycatch of small-scale fisheries in Peru showed that bycatch might be a major threat to sea turtles in the Pacific (Alfaro-Shigueto *et al.*, 2011).

Wilcox *et al.* (2013) created a predictive model to identify areas of higher risk of sea turtle entanglement in ghost nets using estimated densities of sea turtles and ghost nets in the Gulf of Carpentaria, northern Australia. The calculations of at-sea densities of sea turtles were based on prawn trawl fishery's bycatch records. In order to predict the spatial distribution of ghost nets, the authors generated simulated drift pathway of lost nets using an ocean current model (Bluelink Ocean Data Assimilation System), coupled with the results of aerial and beach debris surveys. The results showed increased risk of entanglement along the eastern coast of the Gulf of Carpentaria.

In addition to the usually reported floating debris in the ocean, sea turtles are exposed to the hazards of beach debris as well. Debris stranded in large amounts on sea turtle nesting beaches might have serious effects on the hatchlings (Triessnig *et al.*, 2012). The crawling phase is critical for hatchlings and it has been documented that they often become entangled or trapped in beach debris. Triessnig *et al.* (2012) conducted experiments using 4 different types of debris: empty plastic bottles, Styrofoam cups, plastic canisters and fishing nets. All 199 hatchlings used in the experiment came into contact with debris items and most of them were trapped in cups and canisters and entangled in the nets. They were only able to avoid plastic bottles. Cut-open canisters (containers cut in half) were the most hazardous type of debris, where 84% of hatchlings were trapped.

There has been one report on entanglement of a marine reptile other than a sea turtle. Udyawer *et al.* (2013) found a sea snake (*Hydrophis elegans*) entrapped in a ceramic washer in Queensland, Australia. The ceramic ring circumscribed the animal's body preventing it from ingesting food, which eventually lead to death by starvation. This report is the first record of mortality in sea snake due to entanglement in marine debris.

### *1.7.2.3. Marine mammals*

Almost half of known marine mammal species (52/115) have a propensity to entangle in marine debris (Secretariat of the Convention on Biological Diversity and the Scientific and Technical Advisory Panel – GEF, 2012). According to the number of reports, the most affected group seems to be the pinnipeds, followed by cetaceans. Evidence of entanglement of sirenians and other marine mammals groups is either limited or absent.

Reports on entanglement of pinnipeds are far more common than on other marine mammals. There are two main types of entanglement debris: fishing gear and packaging bands. The species which are particularly affected by entanglement into fishing gear are: Australian sea lion (*Neophoca cinerea*) (Page *et al.*, 2004), New Zealand fur seal ((*Arctocephalus forsteri*) (Page *et al.*, 2004; Boren *et al.*, 2006) Hawaiian monk seal (*Monachus schauinslandi*) (Henderson, 2001), grey seal (*Halichoerus grypus*) (Allen *et al.*, 2012), Antarctic fur seal (*Arctocephalus gazella*) (Arnould and Croxall, 1995; Hofmeyr *et al.*, 2006), Cape fur seals (*Arctocephalus pusillus*) (Shaugnessy, 1980), Californian sea lion (*Zalophus californianus*), northern elephant seal (*Mirounga angustirostris*) and harbour seal (*Phoca vitulina*) (Stewart and Yochem, 1987; Hanni and Pyle, 2000). Four of these species are species of conservation concern by IUCN Red List and include the critically endangered Hawaiian monk seal, the endangered Australian sea lion, the vulnerable northern fur seal and the near threatened Stellar sea lion.

Henderson (2001) investigated the trends in entanglement rates of the Hawaiian monk seals in respect with the implementation of MARPOL Annex V in 1989. The analysed data on seal entanglement in the northwestern Hawaiian Islands spanned from 1982 to 1998. The author did not find changes in occurrence of entanglement before and after the implementation of Annex V. Of 173 entangled animals, 28 seals exhibited injuries caused by entanglement, while 7 individuals were found dead. The field personnel managed to disentangle 118 seals. Juveniles were mainly entangled in nets and adults in monofilament line. The author also confirmed that majority of all entangled individuals (80%) were pups and immature seals, which is in accordance with findings from other authors as well (Arnould and Croxall, 1995; Hanni and Pyle, 2000; Hofmeyr *et al.*, 2006; Raum-Suryan *et al.*, 2009; Waluda and Staniland, 2013).

The annual entanglement rate of Hawaiian monk seals was fairly high (0.7%) compared to some other reported species (Henderson, 2001). For example, Stewart and Yochem (1987) reported entanglement rates in California of: Californian sea lion 0.14-0.22%, northern elephant seal 02%, harbour seal 0.11% and northern fur seal 0%. Hofmeyr *et al.* (2006) found quite low entanglement rates of Antarctic fur seal (0.024-0.059%) in Bouvetoya Island in the South Atlantic. However, higher rates were found in Australian sea lion (1.3%) and New Zealand fur seal (0.9%) on Kangaroo Island, South Australia (Page *et al.*, 2004), New Zealand fur seal (0.6-2.8%) at Kaikoura, New Zealand (Boren *et al.*, 2006), and grey seals (3.6-5%) in Cornwall, UK (Allen *et al.*, 2012). Nevertheless, Henderson (2001) noted that entanglement causes mortalities in Hawaiian monk seal population and that it might have significant impact on some subpopulations, namely from Lisianski Island.

New Zealand fur seals from Kaikoura, New Zealand (Boren *et al.*, 2006), and Australian sea lions and New Zealand fur seals from the Kangaroo Island in South Australia (Page *et al.*, 2004), demonstrated among the world's highest entanglement rates. In 10 years, 185 entangled seals were reported to the

Department of Conservation in Kaikoura, 71 animals were disentangled and released, while 4 were dead. The entanglement rates for these seals were estimated at around 2.84% of the population annually. Debris items, which the seals were most commonly entangled in, were green trawl net (42%) and blue plastic strap (31%). Seals and sea lions from the Kangaroo Island were also mainly entangled in nets and strapping bands, however, there was a difference in prevalence of entangling debris items between species; Australian sea lion was most frequently entangled in monofilament gillnet probably originating from the local shark fishery, while New Zealand fur seals tended to entangle more often in loops of packing band and trawl net remnants from the regional rock lobster and trawl fisheries. It should be noted here that the incidence of bycatch ('ensnaring of non-target animals in active fishing apparatus' Simmonds, 2012, p. 2) of these fisheries might be mistaken for entanglement in marine debris. Entanglement rates in 2002 for each species were 1.3% and 0.9%, respectively. The authors provided an estimate of 1,478 seal mortalities annually in Australia due to entanglement (Page *et al.*, 2004).

Plastic packing straps were reported in several papers as the most frequent entangling object. Two studies on Antarctic fur seals from Bird Island, South Georgia, showed that these animals mainly suffer from entanglement in 'lethal collars' made of packing plastic straps or rubber bands. Croxall *et al.* (1990) removed 170 neck collars from a total of 208 entangled seals between 1988 and 1989. The plastic bands were the dominant item (59%), followed by nylon strings (16%) and fishing nets (13%). Entanglement caused obvious injury in 30% of entangled seals. Waluda and Staniland (2013) carried out a long-term study at the same location, from 1989 to 2013. They documented 1033 entangled individuals of which 791 (77%) were successfully freed. The proportion of plastic straps, synthetic line and fishing nets was 43%, 25% and 17%, respectively. Summer entanglements were more common (64%), than during the winter (36%). There was a significant reduction in entanglement cases after 1994, with the rates of approximately 0.016% per year of the seal population at Bird Island. Higher entanglement rates (0.24%) were reported from Alaska and British Columbia for near threatened Stellar sea lions (386 individuals from 2000-2007), which were also mainly found with plastic packing straps (54%) or rubber bands (30%) around their necks, followed by rope (7%), nets (7%) and monofilament line (2%) (Raum-Suryan *et al.*, 2009).

In two recent reviews of ingestion of and entanglement in marine debris by cetaceans, the authors stated that entanglement of cetaceans is not as well documented as ingestion of debris (Simmonds, 2012; Baulch and Perry, 2013). Baulch and Perry (2014) noted that entanglement has been reported for 14 cetacean species with 73 entanglement incidences, while ingestion of debris affects 48 cetacean species. Sadove and Moreale (1990) found 99 entangled individuals belonging to 17 cetacean species from 1979 to 1988 in New York Bight. Most commonly reported cetacean species with known to entangle include: minke whale (*Balaenoptera acutorostrata*), humpback whale (*Megaptera novaeanglie*), North Atlantic right whale (*Eubalena glacialis*), fin whale (Balenoptera physalus), sperm whale (*Physeter macrocephalus*), pygmy sperm whale (*Kogia breviceps*), bottlenose dolphin (*Tursiops truncatus*), striped dolphin (*Stenella coeruleoalba*) and harbour porpoise (*Phocoena phocoena*) and long-finned pilot whale (*Globicephala melas*) (Sadove and Moreale, 1990; Wells *et al.,* 1998; Knowlton and Kraus, 2001; Johnson *et al.,* 2005; Moore *et al.*, 2009; Cassoff *et al.*, 2011). According to IUCN Red list, North Atlantic right whale and fin whale are endangered, while the sperm whale is considered vulnerable.

In regards to entanglement of sirenians, Beck and Barros (1991) reported 11 manatees lethally entangled in lines and nets in Florida from 1874 to 1985. They also noted that some manatees had scarred or missing flippers caused by entanglements, as well as debris still snarled around the animals. Kikutchi *et al.* (2011) performed an experiment on captive manatees investigating their behavior and the likelihood of entanglement in a net obstacle. They observed that the animals more frequently entangled at night than during the day.

### *1.7.2.4. Fish*

Entanglement of fish in marine debris has not been frequently reported for fish. Out of 16,754 known fish species 66 were reported entangled in marine debris (Secretariat of the Convention on Biological Diversity and the Scientific and Technical Advisory Panel – GEF, 2012). Apart from the species mentioned above as victims of experimental and actual ghost fishing, there are several reports describing entanglement in sharks.

In a long-term study on sharks caught in protective nets off the coast of South Africa from 1978 to 2000, out of 28,687 sharks 53 (0.18%) individuals, belonging to 8 species, were entangled in plastic strapping bands. The most affected species was dusky shark (*Carcharhinus obscurus*) with 27 entangled individuals (0.47%) out of 5,736 caught in that period, with an increase over time. Other shark species found entangled in this study were great white (*Carcharodon carcharias*), sandbar (*Carcharhinus plumbeus*) and blacktip shark (*Carcharhinus limbatus*), with entanglement rates of 0.59% (5/850), 0.38% (2/524) and 0.36% (9/2527), respectively. Dusky, great white and sandbar sharks are listed as vulnerable by IUCN Red list, while the blacktip shark is deemed near threatened.

Degange and Newby (1980) found a large ghost net in the north Pacific which, along with 99 seabirds, entrapped 1 ragfish (*Icosteus aenigmaticus*), 2 salmon sharks (*Lamna ditropis*), more than 200 chum salmon (*Oncorhynchus keta*) and silver salmon (*Oncorhynchus kisutch*). Sazima *et al.* (2002) reported entanglement of sharpnose sharks (*Rhizoprionodon lalandii*) in plastic rings in Sao Paulo, SE Brazil. A recent study on the effects of entanglement on an adult short-finned mako shark (*Isurus oxyrinchus*), vulnerable according to IUCN, showed that a rope wrapped around the body, behind the head, caused a series of injuries and adverse health condition, including a deep lesion in the skin and flesh on the back (8cm wide and 4cm deep), severe scoliosis and malnourishment.

### *1.7.2.5.. Abrasion and smothering of sessile organisms*

Derelict fishing gear is known to affect bottom-dwelling sessile invertebrates. Monofilament lines can cause abrasion and damage to coral colonies that leads to a partial or complete mortality. For example, Asoh *et al.* (2004) and Yoshikawa and Asoh (2004) studied the impacts of fishing lines on cauliflower corals (*Pocillopora meandrina*) in Hawaiian Islands. In 1998 they found that 65% of coral colonies had fishing lines on their surfaces. Mortality occurred with 64% of the colonies, while 17% of corals were partially dead. Colonies which contained fishing line exhibited higher mortality than the ones without it (Yoshikawa and Asoh, 2004). The same results were obtained in 2002, when the authors found higher percentage of dead or damaged corals in fished than unfished zone, as well as higher mortality and damage in coral colonies affected by fishing lines (Asoh *et al.*, 2004).

Similarly, Chiappone *et al.* (2005) conducted a study on 63 offshore coral reefs and hard-bottom sites in Florida Keys National Marine Sanctuary. They found marine debris in 92% of the sites with 87% prevalence of hook-and-line fishing gear, followed by lobster trap gear (10%). Marine debris was observed in contact with 321 sessile invertebrates of which gorgonians (55%), milleporid hydrocorals (17%), sponges (13%), scleractinian corals (30.9%) and colonial zoanthid (*Palythoa mammillosa*) were most affected. Of all hook-and-line fishing gear, 51% caused physical damage to 272 organisms. The most common type of injury was tissue abrasion causing partial individual or colony mortality.

Richards and Beger (2011) examined the detrimental effects of household-related marine debris on coral communities in Majuro Lagoon, Republic of the Marshall Islands. There was a significant negative correlation between the coral and marine debris cover, as well as decreased coral diversity at the site with the greatest debris density.

A survey on marine debris impacts on deep-water habitats and associated benthic communities in Ligurian and Tyrrhenian Sea, Mediterranean, revealed that the local professional and recreational fishing activities have a substantial impact (Bo *et al.*, 2014). Underwater video analysis showed between 19% and 62% of the video frames with an impact to benthic fauna, mainly in a form of entanglement and smothering of corals.

### *1.7.3. Changes in the structure of marine communities*

#### *1.7.3.1. 'Biological pollution' via anthropogenic flotsam*

There are several vehicles which transport non-indigenous species in the marine environment. These include ballast water and solid ballast, hull fouling in commercial shipping and recreational boating, accidental or intentional release associated with the aquarium trade, aquaculture, diving (Bax *et al.*, 2003), marine megafauna (Barnes *et al.*, 2004), as well as floating marine natural and man-made debris (Barnes, 2002; Gregory, 2004; Carson *et al.*, 2013b; Goldstein *et al.*, 2014). The potential of anthropogenic flotsam, as a long-lived substrate, to accommodate and transport various invasive species, as well as change distributional patterns, was already recognised by Carpenter and Smith (1972) and Venrick (1973). It was even suggested by some authors that this might be a positive phenomenon (Wilber, 1987).

All floating objects have the ability to transport marine organisms. Naturally occurring flotsam includes: seeds, wood, floating algae and other plants, animal remains, shells and volcanic pumice (Thiel and Gutow, 2005a; Barnes *et al.*, 2009). Compared to most of these objects, recently introduced synthetic materials in the oceans provide a much more durable 'raft' which may pass longer distances than biodegradable objects, excluding pumice and shells which are also abiotic substrata (Thiel and Gutow, 2005a).

Detailed description of rafting communities is provided in reviews by Thiel and Gutow (2005a, b). They reported 1,205 species which occupy this ecological niche, including cyanobacteria, algae, protists and invertebrates belonging to most marine phyla, as well as some terrestrial taxa. The most common rafting taxa are hydrozoans, bryozoans, crustaceans and gastropods. Many of these organisms are cosmopolitan due to their dispersal via ocean circulation; however some species have local or regional distribution and their occurrence elsewhere is considered a potential threat to the local marine communities (Gregory 2009). Barnes (2002) estimated that floating marine debris doubled the spreading of organisms in the subtropics and tripled it in the higher latitudes. Thiel and Gutow (2005b) stated that some of the rafting organisms are obligate rafters, but that the majority are in fact facultative rafters, and these are mainly benthic dwellers. Aliani and Molcard (2003) collected floating debris from Ligurian Sea, Mediterranean, and found 22 macrobenthic species in 14 samples, mainly molluscs, polychaetes and bryozoans.

Some rafting organisms are of particular concern for the health of generally susceptible coral reef communities. Gold{Pham, 2012 #747}stein *et al.* (2014) examined 242 floating debris items from eastern and western North Pacific. They identified 95 taxa from 11 phyla, with Arthropoda as dominant phylum, followed by Mollusca and Cnidaria. One quarter of all taxa has been recorded in a rafting community for the first time. Potentially invasive species were also confirmed, such as foliculinid ciliate (*Halofoliculina* spp.), a coral pathogen which causes skeletal eroding band (SEB) disease. It was previously documented residing on floating wood, but not plastic objects. Plastic has the potential to spread viruses as well, such as viral haemorrhagic septicaemia virus (Pham *et al.*, 2012) and algae which can cause harmful algal bloom, such as some dinoflagellates (e.g. *Ostereopsis* sp., *Coolia* sp. or *Alexandrium taylori*) (Maso *et al.*, 2003). Pham *et al.* (2012) reported that viruses can remain viable on plastic objects for up to 10 days.

The diversity of microscopic life on floating plastic debris was studied by Zettler *et al.* (2013). They examined plastic pieces collected from the northern Atlantic by Sea Education Association vessel using neuston nets. All plastic pieces were smaller than 5mm. SEM microscope analysis revealed rich microbial community, named ‘plastisphere’, composed of diverse eukaryotic and bacterial microbiota (bacteria, cyanobacteria, diatoms, dinoflagellates, parsinophytes, rhodophytes, etc.). An interesting find of cells (~2μm) embedded in dents on the surface of plastic fragments indicates biodegradation (Fig. 1.7.11.). The cells were not identified. The authors also found various species of bacteria *Vibro*, some of which are human and animal pathogens.

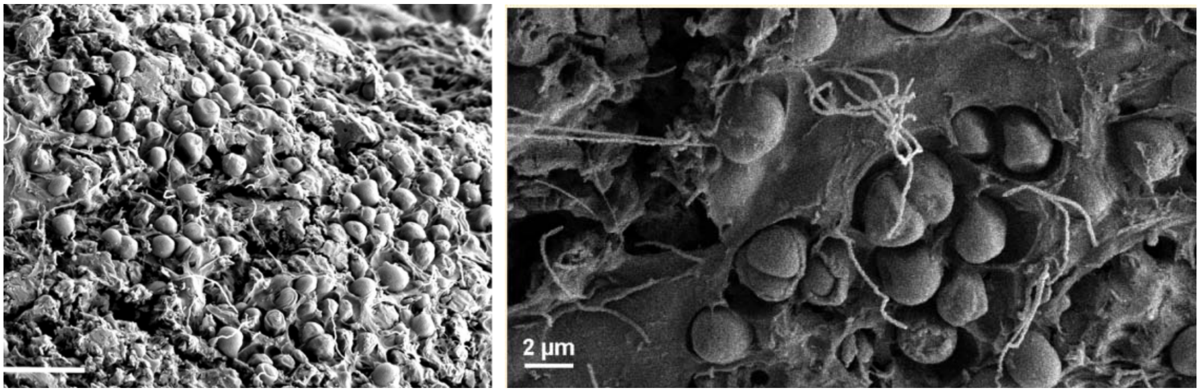

Fig 1.7.11. Microbial cells pitting the surface of plastic of a plastic fragment (indication of degradation) (Zettler *et al.*, 2013).

### *1.7.3.2. Changes in benthic communities*

Marine debris on soft—sediment substrata provides a shelter for benthic organisms such as the octopus (*Octopus vulgaris*) (Katsanevakis and Verriopoulos, 2004), or hard substrata for the settlement of sessile organisms (Katsanevakis *et al.*, 2007). Katsanevakis and Verriopoulos (2004) investigated the relationship between the distribution of the octopus on the soft-bottom sediments and availability of dens, be they a vertical hole in the sediment, rock or stone for building a cavity under it, empty shell or a man-made object. Their experiments demonstrated that the availability of den construction material does influence the occurrence of octopus in the soft sediments. In areas without solid materials, there were no octopi, while at sites with the available construction material, most of 344 octopi used man-made objects as shelters.

Analyses of video records, captured by remotely operated vehicle in Monterey Bay, San Francisco area, at depths up to 3971m, showed that, although some debris caused entanglement, most interactions between debris and marine organisms involved using marine debris as habitat (Schlining *et al.*, 2013). Katsanevakis *et al.* (2007) and Akoumianaki *et al.* (2008) studied changes in benthic community structure, abundance and diversity in relation to the available man-made debris. Both studies found increase in the abundance and diversity with increased number of debris items, as well as compositional changes in community structure. Katsanevakis *et al.* (2007) found ‘increasing positive effect’ for some species, such as gastropod *Hexaplex trunculus*, hermit crab *Cestopagurus timidus*, ascidian *Ciona intestinalis* and the sponge *Chondrilla nucula*. For example, *H. trunculus* used debris to attach egg masses, which caused sharp increase in their abundance. Barnacles *Chthamalus stellatus* used debris for settlement. New intraspecific and interspecific relationships were observed in new changed communities: competition for shelter and hard substratum (e.g. *C. intestinalis*) and predation (*H. trunculus* and *C.*

*stellatus*). Akoumianaki *et al.* (2008) found a 67% change in species composition before and after littering experiment. They also observed significant increase in the abundance of opportunistic polychaetes.

Marine debris covering the sea floor, particularly plastic sheeting, can cause hypoxic and anoxic conditions by inhibiting gas exchange between the sediment and water (Goldberg 1994, 1997). This can affect soft-sediment communities.

#### *1.7.3.3. Changes in intertidal communities*

Beach debris may smother or provide shelter for various organisms living in and of the surface of the sediment. Uneputty and Evans (1997) found decreased abundance of diatoms in litter impacted areas, whereas the abundance of meiofauna was positively correlated with litter presence. Litter cover probably smothered diatoms, while providing shelter and possibly increased decomposition of organic matter (food waste, feces, decomposition of diatoms) and production of bacteria, which enhanced favourable conditions for meiofauna. The abundance in macrofauna was not different between littered and litter-free areas, however the community structure was changed. Polluted areas contained more decapods crustaceans and oligochaetes, while in litter-free areas were dominated by Nereid and spionid polychaetes.

Aloy *et al.* (2011) studied changes in feeding activity of gastropods *Nassarius pullus* in the Philippines. Their experiments showed that beach debris decreases feeding and increases self-burial, as well as causes decrease in the abundance of gastropods during periods of high contamination of the beach.

### *1.7.4. Toxicity of plastics*

Some of the concerns regarding the toxicity of plastic marine debris were mentioned in previous chapters and they are mainly related to ingestion of plastics and subsequent leaching of additives or desorption of adsorbed toxic substances. These toxic chemicals include persistent organic pollutants and plasticisers: polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), petroleum hydrocarbons, organochlorine pesticides, polybrominated diphenylethers, alkyphenols and bisphenol A (BPA), which are either i) added to synthetic polymers during production or ii) attached to the surface of plastic marine debris from the surrounding sea water due to their hydrophobic nature (Teuten *et al.*, 2009; Thompson *et al.*, 2009a). In other words, marine plastic debris acts as both a source and a sink for toxic chemicals (Engler, 2012).

Persistent organic pollutants (POPs) are a wide group of organic compounds with a variety of effects on humans and wildlife. Some additives in plastic (plasticisers) are considered persistent compounds as well (e.g. PBDEs, Darnerud *et al.*, 2001). More detailed information on the types, sources and effects of plasticisers and persistent organic pollutants can be found in Jones and de Voogt (1999), Meeker *et al.* (2009), Oehlmann *et al.* (2009), Teuten *et al.* (2009) and Engler (2012).

Sorption of toxic chemicals onto plastic debris and potential of leaching additives was noted by Carpenter *et al.* (1972) and Carpenter and Smith (1972). They found PCBs on the surface of plastic pellets collected from the Sargasso Sea. It is, however, confounding whether the PCBs were adsorbed from the surrounding water (Carpenter *et al.*, 1972) or there are contained as plasticisers in plastic (Carpenter and Smith, 1972). Mato *et al.* (2001) found PCBs, DDE and nonylphenols on marine plastic pellets from Japanese coast and conducted field experiments to test the possibility of adsorption of these chemicals onto polyproplylene resin pellets. Virgin plastic pellets were deployed for 6 days in the sea water in the industrial complex in Tokyo Bay. The results showed there was a steady increase of PCBs and DDE concentrations (i.e. adsorption) during the experiment, however, the concentrations after 6 days did not reach the levels found on marine plastic pellets - they were 2 orders of magnitude lower. Adsorption coefficient of chemicals to pellets was $10^5$-$10^6$, compared to the concentration in the sea water. Moore *et al.* (2005) did not find PCBs or DDE in their samples of marine plastic pellets from Los Angeles area, however they confirmed PAHs and phthalates in all samples. Rios *et al.* (2007) analysed pellets from the North Pacific Subtropical Gyre, California, Hawaii and Guadalupe Island, Mexico, and detected PCBs, DDTs, PAHs and aliphatic hydrocarbons.

There are a number of other reports on contamination of marine plastic pellets and other plastic debris by persistent organic pollutants and other toxic chemicals. Most commonly reported chemicals include PCBs, DDT and DDE, PAHs, HCH, PBDE, BPA, nonylphenols, alkyphenols, chlordanes and hopanes (Endo *et al.*, 2005; Ogata *et al.*, 2009; Hirai *et al.*, 2011; Heskett *et al.*, 2012; Van *et al.*, 2012; Ryan *et al.*, 2012; Fisner *et al.*, 2013; Mizukawa *et al.*, 2013). Hirai *et al.* (2011) studied POPs and plasticisers associated with plastic debris collected from urban and remote areas in the Pacific and Caribbean waters. They demonstrated that PCB and PAH concentrations were greater in urban beaches than remote and open beaches, most likely due to industrial activities, while DDT concentrations were greater in areas where DDT is still used for controlling malaria (Costa Rica, Vietnam). PCBs are industrial chemicals with a variety of applications and sources, and PAHs are associated with various anthropogenic activities, mainly combustion processes (crude oil and coal). DDT is an insecticide which is no longer in use in agriculture; however it is still used in some areas affected by malaria. PBDEs (flame retardants) and alkyphenols (industrial cleaning agents or plasticisers) were found in all or almost all samples, including those from remote areas. They are either constitutional parts of the plastic debris or were adsorbed from the surrounding water. BPA (plasticiser) was detected only in trace concentrations, most likely due to its lower hydrophobicity compared to other toxic chemicals.

Synthetic polymers are not inert materials, thus have the potential to release their additives and constitutional monomers, before (e.g. marine debris in the ocean) or after the disposal (landfill) (Teuten *et al.*, 2009). The migration of an additive in a polymer structure depends on the sizes of the additive molecules and pores in the polymer, and associated environmental conditions. Most common additives are plasticisers such as phthalates and BPA, as well as stabilisers (organotin compounds), flame retardants (PBDE), UV stabilisers, colorants, biocides and many others (Moore, 2008; Teuten *et al.*, 2009; Thompson *et al.*, 2009a). BPA is, for example, an additive in PVC, a building block of polycarbonate plastics or an additive in printer ink (Teuten *et al.*, 2009). Furthermore, virgin plastic pellets may contain fewer additives than post-consumer plastics as the additives are added during moulding.

Plastics can act as a source of, or a sink for metal contaminants as well. Nakashima *et al.* (2012) examined leaching of toxic metals from beach plastic debris in Goto Islands, Japan. They found chromium (Cr), cadmium (Cd), tin (Sn), antimony (Sb) and lead (Pb) in beach plastics. They noted that PVC fishing floats contained the greatest concentrations of Pb, which exhibited leaching during the leaching experiments. On the other hand, Graca *et al.* (2013) indicated that Styrofoam debris is also a vector for Pb contamination in marine ecosystems, as it readily adsorbs Pb from the water. Holmes *et al.* (2012) conducted adsorption experiments and showed that virgin PE pellets adsorbed metals (Cr, Co, Ni, Cu, Zn, Cd, Pb) from the surrounding sea water, however the concentrations were significantly lower than those found on beached pellets from southwest England coast.

Rochman *et al.* (2014) investigated the sorption of metals (Al, Cr, Mn, Fe, Co, Ni, Zn, Cd, Pb) onto 5 different polymer types (PET, LDPE, HDPE, PVC and PP) over different exposure times (1, 3, 6, 9 and 12 months). The experiments were carried out in San Diego Bay using virgin plastic pellets immersed in the sea water. The experiment demonstrated the increase in the concentration of adsorbed metals over time. All 9 metals were found on all 5 polymer types, excluding Cd on HDPE. There was no significant difference in adsorption potential of different polymer types for these metals.

Conversely, in respect with organic compounds, different polymer types show different adsorption rates (Teuten *et al.*, 2007; Rochman *et al.*, 2013b,c). For example, concentrations of PAHs and PCBs in experimental conditions were recorded consistently greater on LDPE, HDPE and PP, than on PET and PVC. Furthermore, polystyrene (PS), which is considered one of the most unsafe polymers, exhibited greater adsorption potential for PAHs than for any of other polymer types (LDPE, HDPE, PP, PET, PVC) (Rochman *et al.*, 2013c). Polystyrene and expandable polystyrene (Styrofoam) are widely used as food packaging. Saido *et al.* (2014) collected water and sand samples from various locations around Japan and Korea, which were analysed for styrene monomer (SM), dimmers (SD) and trimer (ST), as products of PS degradation. Expanded polystyrene is commonly used by fisheries in this region, as containers and buoys. The results showed higher concentrations of SM, SD, ST in sand samples than in the sea water at the same location. The authors suggest that the toxic chemicals bind to the sand particles where they may remain for long periods.

Desorption of chemicals from plastics has been studied by several authors. Teuten *et al.* (2007) found that sorption rates of organic contaminants (phenantrene) were greater on plastics than on sediment particles. Accordingly, desorption of contaminants was faster from the sediment particles than from plastic. Endo *et al.* (2013) carried out long-term experiments (128 days) investigating desorption of PCBs from PE pellets. They found that the desorption rate depended on the size and complexity of the molecules, as well as the medium. For example, when methanol was added, desorption rate was increased. Bakir *et al.* (2014) studied desorption of contaminants (DDT, perfluorooctanoic acid, phenanthrene, phthalate) from PVC and PE. Plastic particles, presorbed with contaminants, were subjected to different conditions of the sea water and simulated gut surfactants with variable pH and temperature, representing cold and warm blooded animals. Desorption was enhanced in simulated gut conditions, up to 30 times greater than in the sea water, as well as simulated under warm blooded conditions.

Persistent organic pollutants and additives to plastics have various effects on wildlife, some of which include immune and endocrine disruption and mutagenic processes (Oehlmann *et al.*, 2009). The field of ecotoxicology focuses on bioaccumulation and biomagnification of persistent toxins and their effects on individuals, population and the food web level (Teuten *et al.*, 2009). Bioaccumulation occurs within an individual as a result of intake of toxic chemicals which persist in the individual's tissue, while biomagnification refers to accumulation of chemicals through the food web (i.e. ingestion of contaminated food). Plastic ingestion, on a micro- and macroscopic level, is one possible route of accumulation of toxic compounds (Teuten *et al.*, 2009).

Ryan *et al.* (1988) found a positive relationship between the amounts of PCBs, DDT, DDE and dieldrin in fat tissue and eggs of Great Shearwaters (*Puffinus gravis*) and plastic load. They also found a negative correlation between PCB concentrations and the mass of abdominal fat. Jones *et al.* (1996) suggest that increased levels of PCBs, dioxins and furans in Laysan and Black-footed Albatrosses from Midway Atoll might be a result of plastic ingestion, particularly from partially burned and melted plastic pieces.

Bond and Lavers (2011) found elevated concentrations of metals in breast feathers of Flesh-footed Shearwaters from New Zealand (Kauwahaia and Lady Alice Islands) and Australia (Lord Howe Island, Woody Island and bycatch from King George Sound). Metals such as Pb, Cd, Hg and As are used as additives in plastic manufacturing (colourants) and were thus proposed to be elevated due to plastic ingestion. Tanaka *et al.* (2013) found PBDEs (flame retardants), which do not occur in natural pray, in 3 of 12 Short-tailed Shearwaters and implied that ingested plastic might be the possible source of these chemicals in birds tissues.

Rochman *et al.* (2013a) completed a set of experiments on ingestion of LDPE pellets by adult Japanese medaka fish (*Oryzias latipes*). The authors used 2 types of pellets: 'marine' pellets which were kept in the ocean water of Coronado, California, for 3 months, and 'clean' virgin pellets. Pellets were ground to <0.5mm and fed to the fish together with non-plastic ingredients for 2 months. Furthermore, the authors analysed water, plastic, diet and tissue samples for PAHs, PCBs and PBDEs, and examined histopatholgical features of the fish livers. Fish which were fed by 'marine' pellets displayed concentrations of PAHs, OCBs and PBDEs 2.4, 1.2 and 1.8 times greater, respectively, than the fish from the negative control treatment, however, this difference was not evident after 1 month exposure. Authors suggest that short-term exposure to a 10% plastic diet might not present a significant source of contaminants for marine biota. Furthermore, hepatic stress also occurred with fish fed by plastic, including glycogen depletion, fatty vacuolation and single cell necrosis.

Furthermore, Rochman *et al.* (2014) investigated contamination levels of the sea water and fish tissue, (family Myctophidae, lanternfish), which were collected from the predicted South Atlantic subtropical gyre (west from South Africa), together with trawl samples to determine the density of surface plastic debris. Firstly, the authors did not find significant difference in debris density among different sampling zones of the gyre, predicted by previous authors. Secondly, water samples showed contamination only by PCBs and PBDEs, but not by BPA, aklylphenols and alkylphenol ethoxylates, however fish tissue did contain all 5 chemicals. BPA levels were greatest in areas with smallest debris densities; while the tissue levels of alkyphenol ethoxylates were greater in fish from greater plastic concentration areas. For other contaminants, no significant difference was observed between locations. Further examination showed significant relationship between the levels of PBDEs and individual stations within locations, where greater PBDE concentrations were documented at stations with greater debris densities.

### *1.7.5. Impacts on humans*

#### *1.7.5.1. Health impacts*

The threat of marine plastic debris to human health can be direct, such as from physical injuries or infections by pathogenic bacteria or viruses, or indirect, through contamination through ingesting contaminated food (Sheavly and Register, 2007; Williams *et al.*, 2013). Physical harm is related to beach debris and sharp objects (e.g. broken glass, knives and needles), submerged objects and entanglement in by swimmers, snorkelers and divers, and floating debris and at sea collisions (Sheavly and Register, 2007; Moore, 2008; Williams *et al.*, 2013). Moore (2008) noted that, according to NOAA's office of Response and Restoration, in 2005 there were 269 boating accidents caused by floating and submerged marine debris, which resulted in '15 deaths, 116 injuries and $US 3 million in property damage' (p. 132). Cho (2005) reported that 292 lives were lost in a severe maritime accident in 1993 in Korea due to entanglement of fishing ropes around the shafts and the propeller. Furthermore, medical and sanitary waste (e.g. infected syringes, condoms, tampons, pads, nappies) can carry pathogens and cause various infections and conditions, such as infectious hepatitis, diarrhea, bacillary dysentery, skin rashes, typhoid and cholera (Sheavly and Register, 2007).

Numerous studies have demonstrated that sea organisms ingest plastic debris. Microplastic is usually ingested by invertebrates and macroplastic objects by organisms of higher orders. Since plastic often contains toxic chemicals, which are released into the tissue of organisms affected by plastic ingestion, consuming these animals as seafood presents an indirect risk from plastic pollution to human health. This particular relationship has not been investigated yet, however, contamination of seafood by plastic and associated toxic compounds has been proven (Murray and Cowie, 2011; Gassel *et al.*, 2013; Mathalon and Hill, 2014), as well as the adverse effects of persistent organic pollutants and additives in plastic to human health (Koch and Calafat, 2009; Schug *et al.*, 2011). One study showed hormonal changes in humans which consumed contaminated fish with elevated PCB and DDE levels from the Great Lakes, USA, however the contamination of fish was not linked to plastic pollution. Nevertheless, accumulation of plastic debris-related contaminants through the food web, would also be a possible indirect harm to human health from plastic pollution.

#### *1.7.5.2. Economic impacts*

Apart from environmental issues, which most published literature is focused on, and human health concerns, marine debris pollution also has a negative economic impact on several important industries, including shipping, fisheries and tourism (UNEP, 2009; Mouat *et al.*, 2010; McIlgorm *et al.*, 2011). McIlgorm *et al.* (2011) estimated the overall cost of marine debris pollution to the 21 economies of the Asia-Pacific rim to be US$ 1.26 billion in 2008. Impacts of marine debris pollution can be placed into one of 3 categories:

- **direct economic cost** – easily measurable costs (e.g. downtime for defouling a propeller entangled)
- **indirect economic impact** – not easily measurable (e.g. plastic ingestion and contamination of food chain)
- **non-market values** – not easily measurable (e.g. impact of marine debris on non-market values such as scenic values)

##### *1.7.5.2.1. Fisheries and boating*

Marine debris pollution mainly impacts the fishing industry through the loss of catch in ghost fishing (directly of commercial species, or indirectly of the prey of commercial species) and through damage to vessels and fishing gear (McIlgorm *et al.*, 2011). Several authors provided estimates of economic loss due to ghost fishing mortality. Al-Masroori *et al.* (2004) conducted a study on damage of ghost fishing of derelict fish traps to the local traditional fishing communities. In a 21-week experiment, estimates of loss in fish traps were made mainly once a week, which certainly underestimates the real loss. Nevertheless, the authors reported 426 fish and 60 cuttlefish (total weight of 434.3kg) caught in 25 fish traps over the course of the experiment. A majority of the catch were commercial fish (83%) and the rest had no commercial value. The estimated economic loss to this particular industry was ~US$145 per 3 months or ~US$168 per 6 months. The economic loss is not linear due to observed decrease in trap ghost fishing mortality over time.

Gilardi *et al.* (2010) estimated and compared the economic loss caused by ghost fishing and the cost of removal of the derelict fishing gear causing mortality in Puget Sound, Washington, US. They estimated that the life span of a derelict gillnet is around 10 years and a daily catch of Dungeness crabs in the net is 1.2, which gives a total of 4,368 crabs and the cost of US$ 19,656. Compared to this loss, removal of derelict nets was calculated to cost only US$ 1,358. Bilkovic *et al.* (2014) estimated the annual loss of about 900,000 crabs and US$ 300,000 to blue crab fisheries in Virginia Waters of Chesapeake Bay, US due to crab pot ghost fishing. They also noted that between 10 and 70% of pots are lost annually and that they have the potential to ghost fish for several years.

Ghost fishing is an unwanted side effect of fishing industry, however fisheries are also affected by non-fishing-related debris, such as plastic bags and packaging (Nash, 1992; Eryasar *et al.*, 2014). Nash (1992) described the difficulties Indonesian artisanal fishermen were facing in their traditional fishing grounds after it had been affected by marine debris pollution originating from the local coastal landfill. As stated by the fishermen, debris caught in the nets makes them visible to the fish. Furthermore, clearing the nets is time-consuming and the repair is costly. Other problems include propeller entanglement, injuries, changing the fishing gear (e.g. from nets to hook and line) and changing location or not fishing due to visible floating debris. Most commonly reported debris items were plastic bags. Eryasar *et al.* (2014) reported that non-fishing-related sea floor debris may affects bottom trawling by blocking the grids in as much as 85% of trawl hauls.

Plastic debris causes damage to vessels by entangling around the propellers, keels and rudders and clogging up the intake ports (Moore, 2008; Mouat *et al.*, 2010; McIlgorm *et al.*, 2011). Mouat *et al.* (2010) noted that marine debris causes between € 11.7 and 13 billion worth of damage to Scottish fishing fleet annually, which 5% of the total industry's profit. Regarding vessel damage, entanglement of propellers in derelict fishing gear is the most commonly reported debris-related incident in UK. In 2008, there were 286 reports and rescues to the vessels due to fouled propellers, with the cost between € 830,000 and 2,189,000 (Mouat *et al.*, 2010).

#### *1.7.5.2.2. Tourism*

There is generally little information on the economic loss to the tourism industry due to marine debris pollution. McIlgorm *et al.* (2011) provided estimated losses for the Asia-Pacific Economic Cooperation (APEC) region. In 2008, an overall loss to all marine economy sectors in APEC region (i.e. fisheries, tourism, marine construction, oil industry, marine services) due to marine debris pollution was estimated at US$ 1.265 billion. Tourism industry alone suffered economic loss of US$ 622,000,000, which is almost 50% of the entire loss caused by marine debris pollution.

The impacts of marine debris mainly relate to the loss of aesthetic value of the beaches by stranded plastic debris and the cost of the cleanups (McIlgorm *et al.*, 2009). McIlgorm *et al.* (2011) noted that shoreline cleanups can range from US$ 100 – 20,000 per collected tonne. Several authors studied public opinion and beach users' perception of the beach quality and they confirmed that marine debris is an important determining factor (Tudor and Williams, 2003; Roca *et al.*, 2009; Quintela *et al.*, 2012). Some case studies showed that isolated marine debris contamination events can have severe impacts on tourism in a certain area (Ofiara and Brown, 1999; Jang *et al.*, 2014). For example, in July 2011 Goeje Island in South Korea was affected by a marine debris pollution event resulting from heavy rains. This event had a direct impact on island's tourism industry, with a 63% reduction in the number of visitors between 2010 and 2011 and a loss of US$ 29 – 37 million (Jang *et al.*, 2014).

## 1.8. Management approach

Marine debris pollution is a diffuse type of pollution associated with an array of human activities on land and at sea, causing numerous adverse effects on the ocean ecosystems and humans. For this reason, management of marine debris requires an integrated approach, encompassing science, legislation, economics and society. Some key strategic activities to reduce the impacts of marine debris were established through the Honolulu Strategy at the Fifth International Marine Debris Conference, held in March 20-25, 2011, in Honolulu, Hawai'i (NOAA and UNEP, 2012). The strategy provides principles, goals and strategies that are applicable globally, regardless of specific conditions or challenges, for reducing the ecological, human health and economic impacts of marine debris. The main goal of Honolulu Strategy is to prevent, reduce and manage land-based and marine-based debris, as well as to remove accumulated marine debris from the coastal and marine environment. Since the problem of marine debris pollution is extraordinarily complex, successful implementation of the Honolulu Strategy will require participation and support on multiple levels – global, regional, national and local – involving the full spectrum of civil society, government and intergovernmental organizations, and the private sectors. Furthermore, identifying the sources of marine debris is an imperative for good marine debris management (UNEP, 2009).

At the same conference in Hawai’i, the plastics industry responded to the efforts of marine debris community to combat this pollution and signed the Declaration of the Global Plastics Associations for Solutions on Marine Litter (Global Declaration). By December 2013, 60 plastics associations representing 34 countries signed the Global Declaration. Furthermore, more than 185 projects are planned, ongoing or completed and include beach cleanups, improving waste management, global research and education and awareness campaigns (The Declaration of the Global Plastics Associations for Solutions on Marine Litter – Progress Report, 2014).

### *1.8.1. Research and monitoring*

To assess the extent of the pollution problem in a given area, including potential sources, public opinion, impacts to wildlife and humans, as well as to define management steps, research and appropriate measurements are needed (Earll *et al.*, 2000). However, the field of marine debris pollution generally does not lack research effort as much as standardization of methods, measurement units and terminology, as well as clarity of methodology description and presented data. Reviewing marine debris literature, in respect with quantitative data of stranded, submerged and floating marine debris, often seems like a ‘Sisyphean task’. The need for standardization of methods has also been pointed out by several other authors (Dixon and Dixon, 1981; Ribic and Johnson, 1990; Ribic *et al.*, 1992; Rees and Pond, 1995; Velander and Mocogni, 1999; Williams *et al.*, 1999; Earll *et al.*, 2000; Williams and Tudor, 2001; Williams *et al.*, 2013), and is particularly true for beach debris studies (Williams *et al.*, 2013).

It should be stressed out that, for comparative studies of beach debris, attention should be given to the following aspects of the sampling procedure and subsequent analyses:

- **size of sampled debris** – for example, data which include debris larger than 0.5cm (Smith and Markic, 2013) are not comparable to data including debris larger than 2cm (Topcu *et al.*, 2013), as the entire debris size range between 0.5-2cm is excluded
- **survey area** – for example, samples of beach debris accumulations on a strand line (Silva-Cavalcanti *et al.*, 2009) will show much greater densities than samples collected from the entire beach area (Hayward, 1999).
- **measurement units** (e.g. item $m^{-1}$ of the beach front vs. item $m^{-2}$ of the beach face)

- **debris categorisation** – for example, some surveys include only plastic (Johnson, 1990), while others may include all types of anthropogenic debris, including organic waste (e.g. food) (Kitto *et al.*, 2011) or 'naturally occurring rubbish' such as dog feces, marine organisms, sea weed and driftwood (Silva-Iniguez and Fischer, 2003)
- **temporal aspect of debris quantity** – some studies provided densities of marine debris based on one-off assessments (Smith, 2012), but there have been cases where debris density represented an average of multiple temporal samples (e.g. over 2.5 years, Storrier *et al.*, 2007), which is considered to represent accumulation rates in other studies (e.g. Eriksson *et al.*, 2013)

Several best practice proposals and guidelines on sampling methods have been proposed by individual authors (Dixon and Dixon, 1981; Ribic *et al.*, 1992; Earll *et al.*, 2000; Ryan *et al.*, 2009), and national and international organisations, such as OSPAR Commission, UNEP, NOAA and European Commission (Ospar Commission, 2007; Cheshire *et al.*, 2009; Opfer *et al.*, 2012; European Commission, 2013). However, consensus at the global level has not yet been realised and various methods are still in use. Some authors pointed out that there is no best sampling method as each study has different objectives (Velander and Mocogni, 1999).

Environmental monitoring is often seen as needless and ineffective, however when properly designed, it delivers multiple benefits (Lovett *et al.*, 2007). It is defined as 'a time series of measurements of physical, chemical, and/or biological variables, designed to answer questions about environmental change' (Lovett *et al.*, 2007, p. 253). Well-planned long-term monitoring, with clear questions and appropriate methodology, should provide the information needed for testing of models and hypotheses and for short-term experiments and observations. It may also present the basis for environmental policy, in both developing the legislation and evaluating its effectiveness, as well as for assessing the efficacy of management measures (Lovett *et al.*, 2007; Ryan *et al.*, 2009).

Monitoring of marine debris should address the following questions (Ryan *et al.*, 2009, p. 2000):

- *What is the abundance, distribution and composition of plastic litter, and are these attributes changing over time?*
- *What are the main sources of plastic litter, and are they changing over time?*
- *What are the impacts of plastic litter (environmental and economic) and are they changing over time?*

Long-term studies of marine debris pollution are discussed and listed in the tables in Chapter 1.6. Large-scale and/or long-term studies of stranded debris are usually carried out as a part of cleanup events with volunteers as data collectors (Dixon and Dixon, 1981; Edyvane *et al.*, 2004; Law *et al.*, 2010) or as a part of a monitoring program (Morshige *et al.*, 2008). Some monitoring methods are not direct measurement of the marine debris in the marine physical environment, but its presence in marine organisms, such as seabirds (Ryan *et al.*, 2009). For example, Avery-Gomm *et al.* (2012) used Northern Fulmars (*Fulmarus glacialis*) for biomonitoring, as indicators of marine debris pollution. Other proposed unconventional methods include webcam beach litter monitoring (Kako *et al.*, 2010) and monitoring of surface debris by SOAR (SmartphOne-base Aquatic Robot), a robotic fish with an incorporated smart phone (Wang *et al.*, 2014).

### *1.8.2. Legislation*

There are two international legally binding agreements, within the framework of the United Nations Convention of the Law of the Sea, relating to the protection of the marine environment from dumping wastes from ships, MARPOL 73/78 (Annex V, International Convention of the Prevention of Pollution from Ships 1973, modified by the Protocol of 1978) and London Convention (Convention of the Prevention of Marine Pollution by Dumping of Wastes and Other Matter 1972, updated by the Protocol 1996) (Criddle *et al.*, 2009). The London Convention has been in force since 1975 and Annex V of MARPOL 73/78 entered into force on 31st December, 1988. MARPOL Annex V addresses the prevention of the pollution by garbage from ships and specifically prohibits at-sea discharge of any type of plastics. Additionally, in 1995, UNEP Secretariat convened an intergovernmental conference at which 108 countries adopted a Global Programme of Action for the protection of the marine environment from land-based activities (GPA) (UNEP, 2009). The associated Washington Declaration (1995) has been renewed through Montreal (2001), Beijing (2006) and Manila Declarations (2012). These instruments aim to regulate, among others, marine debris pollution at a global level.

On a regional level, there are several conventions and directives in relation to marine pollution from ocean and land-based sources, which were developed by UNEP (Regional Seas Programme) and other agencies, such as the Secretariat of the Pacific Regional Programme (SPREP), OSPAR, HELCOM and European Commission, and include:

- Noumea Convention for the protection of the natural resources and environment of the South Pacific Region (1986), with the Protocol for the Prevention of the pollution of the South Pacific Region by dumping (1986) (SPREP, 1986)
- Lima Convention for the protection of the marine environment and coastal area of the Southeast Pacific (1981), with the **Protocol for the Protection of the South-East Pacific against Pollution from Land-based Sources (1983) – UNEP**
- Antigua Convention for cooperation in the protection and sustainable development of the marine and coastal environment of the Northeast Pacific (2002) - UNEP
- Cartagena Convention for the protection and development of the marine environment of the Wider Caribbean Region (1983), with the Protocol on the prevention, reduction and control of land-based sources and activities (LBS Protocol) (1999) - UNEP
- Kuwait Regional Convention for Co-operation on the Protection of the Marine Environment from Pollution (1978), with the Protocol for the Protection of the Marine Environment against pollution from land-based sources (1990) - UNEP
- Barcelona Convention for the Protection of the Marine Environment and the Coastal Region of the Mediterranean (1976), with the Protocol for the Prevention and Elimination of pollution of the Mediterranean Sea by Dumping from Ships and Aircrafts (Dumping Protocol) (1976) and **Protocol for the protection of the Mediterranean Sea against pollution from land-based sources (LBS Protocol) (1980) – UNEP**
- **Helsinki Convention on the protection of the marine environment of the Baltic Sea area (1974 and 1992) – HELCOM Commission (Baltic Marine Environment Protection Commission)**
- OSPAR Convention for the protection of the marine environment of the North-East Atlantic (1992), a joint convention of Oslo and Paris Conventions (Oslo Convention for the prevention of marine pollution by dumping from ships and aircraft, 1972; Paris Convention for the prevention of marine pollution from land-based sources (1972) – OSPAR Commission
- Marine Strategy Framework Directive (MSFD, 2008/56/EC) (2008) – European Commission

Literature on the effectiveness of marine debris legislation and regulation is generally outdated (Bean, 1987; Lentz, 1987; Baur and Iudicello, 1990; Brillat and Liffmann, 1991; Nollkaemper, 1994; Kimball, 1995; Henderson, 2001). Henderson (2001) studied the entanglement of critically endangered Hawaiian Monk Seal in plastic debris and found no change in occurrence before and almost 10 years after the implementation of MARPOL Annex V. It is still emphasized in the literature that more effort should be directed towards the enforcement of the existing treaties.

> '*Unfortunately, laws do not guarantee compliance. In addition to enforcement and penalties, a sense of environmental stewardship among ocean users is essential for laws to be effective. While there are laws regulating the dumping of trash at sea and on shore, the global nature of debris, its inability to be confined within territorial boundaries, and the complexity of identifying debris sources have made effective laws difficult to draft and even harder to enforce.*' – Sheavly and Register, 2007

Although the International Maritime Organization (IMO), the United Nations' specialised agency, develops and provides regulatory framework, there are still many omissions regarding the enforcement of international maritime law in relation to dumping of wastes at sea (MARPOL Annex V). This task is very challenging due to the vast expanse of the oceans and the lack of state authority and control beyond 200 miles offshore, leaving large 'grey' area for undetected discharge of plastic debris (Rakestraw, 2012). The IMO, however, does encourage states to enforce regulations by, for example, improving their port reception facilities. According to MARPOL, port state officers have the authority to inspect a foreign-flagged vessel if there is suspicion that the crew may not be familiar with Annex V regulations. Furthermore, Regulation 9 of Annex V requires that ships over 400 gross tonnes maintain a detailed garbage management plan, including garbage management record books.

Additionally, a number of ships sail under the flags of foreign countries with which they have no or very little relations - 'Flags of convenience' (FOC) (Rakestraw, 2012). It is believed that these ships are the main contributors to marine pollution as they are virtually beyond the reach of law.

> '*Over one third of all ships throughout the world are registered under FOC. The decision where to flag a vessel usually stems from economic rather than political or geographical considerations and FOC states provide less stringent enforcement of environmental regulations, thereby giving ship owners the economic advantage of lower labor rates, relaxed safety standards and lower foreign tax rates. The largest FOC states are Liberia, Panama, Malta, the Bahamas, and Antigua. Of the thirty-two FOC states identified by the International Transport Worker's Federation, ten are listed on the Paris MOU's black list, which indicates a "very high" or "high" risk of inspection violations.*' – Rakestraw (2012)

Academic literature on legislation relating to land-based sources of marine debris pollution (GPA) is practically non-existent. Nollkaemper (1994) provides an insight into the regulation of land-based sources of marine debris from local to global level. Marine debris pollution originating from land is ascribed to wastewater discharges (combined sewer outflows, storm water discharges and lack of sewage treatments), run-off from landfills and illegal dump sites located near rivers or on the coast, and beach littering. These sources of pollution are largely unregulated and the associated national laws in many countries are either absent or inadequate. In comparison to treaties addressing discharge of wastes at sea, 'international rules to induce appropriate national and local policies on land-based discharges lead a marginal existence' (Nollkaemper, 1994, p. 649). This paper dates from right before GPA was adopted, thus the information is outdated.

In January 2012, at the Intergovernmental Review Meeting on the implementation of GPA (Third session) in Manila (Manila Declaration), the issue of land-based marine litter was highlighted as a priority source category for the period 2012-2016. The importance of 'Honolulu Strategy' and Honolulu Commitment' was stressed as well. The Global Partnership on Marine Litter (GPML), guided by the Honolulu Strategy, and the Global Partnership on Waste Management have been established, for further collaboration between international agencies, governments, businesses, academia, local authorities, non-governmental organisations and individuals. The specific objectives of GPML are (UNEP, GPA):

- To reduce the impacts of marine litter worldwide on economies, ecosystem, animal welfare and human health.
- To enhance international cooperation and coordination through the promotion and implementation of the Honolulu Strategy - a global framework for the prevention and management of marine debris, as well as the Honolulu Commitment – a multi-stakeholder pledge.
- To promote knowledge management, information sharing and monitoring of progress on the implementation of the Honolulu Strategy.
- To promote resource efficiency and economic development through waste prevention (e.g. 4Rs (reduce, re-use, recycle and re-design) and by recovering valuable material and/or energy from waste.
- To increase awareness on sources of marine litter, their fate and impacts.
- To assess emerging issues related to the fate and potential influence of marine litter, including (micro) plastics uptake in the food web and associated transfer of pollutants and impacts on the conservation and welfare of marine fauna.

Management of the marine pollution in the South Pacific region is described in detail, although more than a decade ago, by Morrison (1999). The paper provides the history of regional activities relating to environmental management and marine debris pollution programs, as well as the establishment and development of major South Pacific non-government organisations, such as the Pacific Community (SPC) and SPREP. The South Pacific Commission was founded in 1947 and renamed The Secretariat of the Pacific Community in 1997. Its role is to provide advisory services and support in various fields (social, economic and cultural) to 22 governments and administrations of the region. Thereafter, other relevant organisations were established as well, including: South Pacific Regional Environment Programme (SPREP), South Pacific Forum (political and economic cooperation), South Pacific Applied Geoscience Commission (SOPAC), South Pacific Forum Fisheries Agency (FFA), University of the South Pacific (USP) and the Tourism Council of the South Pacific (SPPA).

The program for the protection of the environment of the South Pacific was initiated by a Regional Symposium on the Conservation of Nature, Reefs and Lagoons (Noumea, New Caledonia, 1971). In 1982, SPREP was established at the Conference on Human Environment in the South Pacific in Rarotonga, Cook Islands, as a multy-agency programme (UNEP Regional Seas, UN Economic and Social Commission for Asia and the Pacific (ESCAP), SPC, Forum). The programme lacked trained manpower and resources for the research and control of marine pollution, including equipment, facilities, funding and staff at the time. The Noumea Convention, with associated protocols, came into force in 1990, to address pollution: pollution from ships, dumping, land-based sources, sea bed exploration and exploitation, atmospheric discharges, storage of toxic wastes, testing of nuclear devices, mining and coastal erosion.

### *1.8.3. Improved waste and stormwater management*

Since the majority of marine debris originates from land, a consequence of street and beach littering, runoff from landfills, and untreated discharges of wastewater, improvement of waste and wastewater management is crucial for sound marine debris management. To the writer's knowledge, there are no studies which specifically focus on the relationship between waste management practices and marine debris levels in a certain area.

Sustainability is considered the main concept to guide modern-day waste management practices (Hou *et al.*, 2012). Most common current waste management strategies include landfilling, incineration and waste-to-energy conversion, composting and the so-called 3Rs: reducing, reusing and recycling of plastic (Gusti, 2009; Hopewell *et al.*, 2009; Shekdar, 2009).

Recycling of plastics started in the 1970s (Hopewell *et al.*, 2009). The need for recycling arose from the need for improved waste reduction and resource recovery (Hopewell *et al.*, 2009; Hou *et al.*, 2010). Around 8% of global oil and gas production is used for manufacturing synthetic materials, 4% on the products and 3-4% for the energy to produce them. Most of these materials are either disposable packaging or short-lived plastic products (Hopewell *et al.*, 2009). Recovery rates in developed countries are usually greater than in developing countries. For example, in England in 2010/11 recovery rates of household waste were 40%, and in US 34% in 2010 (Hou *et al.*, 2012). In developing countries, the recovery rates range from 5-41%, namely 5% in Nepal, 7-10% in China, 13% in Philippines, 15% in Thailand, 13-20% in Vietnam and 41% in Brazil (Troschinetz and Mihelcic, 2009). Factors influencing recycling in developing countries are:

- government policy and finance,
- waste characterization, collection and segregation
- household education and economics
- municipal solid waste management (MSWM) administration, personnel education and plan
- local recycled-material market
- technological and human resources
- and land availability

Storm water washes the streets and other surfaces in urban areas, and gathers, among other pollutants, anthropogenic litter as well, and transfers it into the marine environment (Armitage and Rooseboom, 2000; Fitzgerald and Bird, 2011). There are two ways of preventing street litter entering the storm water drainage system: installation of grids over catchpit entrances and reducing the source litter load. The latter should be managed through improvements to urban waste management and anti-littering legislation. Some of the suggested actions for street litter reduction are: improvement of rubbish bin system and street sweeping, organized cleanups and educational campaigns (Armitage and Rooseboom, 2000). Prevention of existing street litter from entering the drainage system by physically blocking its route towards the marine environment can be done using gross pollutant traps (GPTs), which are defined as 'devices used for water quality control that remove solids typically greater than five milimetres conveyed by stormwater runoff' (Fitzgerald and Bird, 2011). They have been used internationally as a primary treatment of storm water in storm water management. This system is quite costly and requires regular maintenance. There are several types of GPTs:

- **drainage entrance treatments**: grate entrance systems, side entry catchpit traps and gully pit traps
- **direct screening devices**: litter collection baskets, release nets, trash racks, return flow baskets and channel nets
- **floating traps**: flexible floating booms, floating debris traps
- **non-clogging screens**: circular and downwardly inclined screens

- **sediment traps**: sediment settling basins and ponds, circular settling tanks, hydrodynamic separators.

A study on the street litter collected from a single litter trap of 7.5 $m^3$ $s^{-1}$ flow rate in Springs, South Africa in 1990/91, showed that the trap had the capacity to retain 186 $m^3$ litter annually from a catchment area of about 299ha (85% commercial/industrial and 15% residential area) (Armitage and Rooseboom, 2000). The dominant litter type was plastic (62%), followed by polystyrene (11%), paper (10%), tin cans (10%), glass (2%) and other items (5%).

Land waste reduction is of great importance for marine debris management, particularly on islands, due to their limited land area. Historically, waste generation on islands did not require complex waste management, however this changed over time with development and the growth of local tourism industries (Periathamby, 2011). Many islands still have poor infrastructure and inadequate waste facilities with limited waste collection, which is associated with the lack of resources, collection vehicles, labor and disposal sites. Islanders mostly dispose of their waste by dumping in a backyard pit (57%) or open backyard dump site (37%) and sometimes into waterways (1%) (Periathamby, 2011). Common methods of disposing collected waste are landfilling and waste incineration. The 3R strategy is successful mainly in more developed islands, such as Guam (Periathamby, 2011). During waste collection or transport rubbish may get lost, blown off or washed away from disposal sites. Almost half a million tonne of municipal solid waste is generated annually in Hawaii and is collected and transported to the mainland (Washington, Idaho, or Oregon) about once a week. The transport increases the risk of inadvertent discharge of rubbish into the ocean due to overload or adverse weather conditions. Additionally, ocean dumping of garbage has been practiced worldwide as well.

UNEP (2011) and Global Partnership on Waste Management issued a Work Plan for marine litter for 2012-2016. The strategies to reduce the marine debris from land-based sources proposed by the Global Parthership are (UNEP, 2011):

- Conduct education and outreach on marine debris and solid waste minimization and management
- Employ market-based instruments to support solid waste minimization and management
- Employ infrastructure and best practices for stormwater and solid waste minimization and management
- Develop and enact legislation and policies to support solid waste minimization and management
- Improve the regulatory framework regarding stormwater, sewage systems, and debris in tributary waterways
- Build capacity to monitor and enforce compliance with regulations and permit conditions regarding litter, dumping, solid waste management, stormwater, and surface runoff
- Conduct targeted cleanup efforts on coastal lands, in watersheds, and in waterways

### *1.8.4. Initiatives and additional measures for prevention and remediation of marine debris pollution*

Prominent international and national organisations addressing the problem of marine debris pollution include UNEP, NOAA, EPA, IMO, IOC (Intergovernmental Oceanographic Commission of UNESCO), GEF (Global Environment Facility) and the European Commission. Initiatives and measures for prevention and mitigation of marine debris pollution are in fact quite numerous (Table 1.8.1.). They include organised volunteer cleanups, education and public outreach, and research.

Table 1.8.1. List of organisations and initiatives worldwide addressing marine debris pollution problem.

| Organisation/Project/ Initiative | Description |
|---|---|
| 5 Gyres | NGO funded by Anna Cummins and Markus Eriksen, expeditions, scientific research, http://5gyres.org/ |
| Algalita Marine Research Institute | NGO founded in 1994 by Charles Moore, extensive scientific work and numerous expeditions, Junk expedition http://junkraft.blogspot.co.nz/ http://www.algalita.org/index.php (Long Beach, California) |
| B.E.A.C.H. (Beach Environmental Awareness Campaign Hawai'i) | NGO, education, debris removal and research, plastic reduction and litter prevention Hawai'I http://www.b-e-a-c-h.org/index.html |
| Clean Seas Coalition | Coalition, California http://www.cleanseascoalition.org/ |
| Cleanup Australia Day and Cleanup the World | Cleanup, community based environmental campaign, since 1990 in Australia and 1993 in the world, http://www.cleanuptheworld.org/en/ |
| Coastcare | Coastcare program established by the Australian Government in 1995, volunteers, Carpentaria Ghost Nets Programme http://www.coastcare.com.au/ |
| Coastwatch Europe | International network of environmental groups, cleanups, scientific research, http://coastwatch.org/europe/ |
| CCAMLR (Commission for the Conservation of Antarctic Marine Living Resources) | Established in 1982, CCAMLR Marine Debris Program, active in research in the Antarctic area, https://www.ccamlr.org/en/science/marine-debris |
| CSIRO (Commonwealth Scientific and Industrial Research Organisation) | 3-year survey on marine debris in Australian waters (TeachWild) , research scientist Denise Hardesty http://www.csiro.au/Organisation-Structure/Flagships/Wealth-from-Oceans-Flagship/marine-debris.aspx |
| Earthwatch | International NGO founded in 1971, Earthwatch Australia developed TeachWild, a 3-year marine debris program in collaboration with CSIRO http://teachwild.org.au/ |
| East Asian Civil Forum on Marine Litter | Forum established in 2009 in Japan, gathers NGOs from E Asian countries: JEAN (Japan Environmental Action Network), OSEAN (Our Sea of E Asia Network), TOCA (Taiwan Ocean Cleanup Alliance), Green Fins Association from Thailand |
| EPA Region 9 (US Environmental Protection Agency) | US Government agency founded in 1970, Marine Debris Strategy: Prevention, Reduction and Cleanup, Trash Free Waters, education and outreach and research http://www.epa.gov/region9/marine-debris/ |
| Expedition MED | EU research program launched in 2010, scientific campaign, research, raising awareness, http://www.expeditionmed.eu/fr/en/ |
| GhostNets Australia | Alliance of indigenous communities founded in 2004, raising awareness and research (N Australia) http://www.ghostnets.com.au/about/ |
| Global Garbage | Founded by Fabiano Barretto, http://www.globalgarbage.org/blog/ |
| International Pellet Watch | Project for global monitoring of POPs using beached plastic resin pellets, Initiated by Hideshige Takada in 2005 http://www.pelletwatch.org/ |
| IPRC (International Pacific Research Center) | Marine debris and ocean current modeling (Hawai'i) http://iprc.soest.hawaii.edu/ |
| Keep Britain Tidy | Independent charity founded in 1960, cleanups and education, UK http://www.keepbritaintidy.org/home/481 |
| Keep New Zealand Beautiful | Established under the New Zealand Litter Act in 1979, nation-wide programs, litter abatement, education, http://www.knzb.org.nz/ |
| KIMO International (Kommunenes Internasjonale Miljøorganisasjon) | NGO founded 1990 in Denmark, activism and research http://www.kimointernational.org/Home.aspx |

| | |
|---|---|
| Legambiente | Clean-up the Med since 1995, Italy http://www.legambiente.it/contenuti/campagne/clean-med |
| Let's do it Mediterranean | Cleanup since 2014, http://ldim.net/en/ |
| MARE (Marine Affairs Research Foundation) | Launched online discussion forum MarineDebris.Info, coordinator Nick Wehner, http://marinedebris.info/ |
| Marine Defenders | Educational Program (W US coast), http://www.marinedefenders.com/ |
| Marine Litter Solutions | Global association of plastic makers founded in 2011, Declaration of the Global Plastics Associations for Solutions on Marine Litter, marine debris projects, http://www.marinelittersolutions.com/ |
| NOAA | NOAA Marine Debris Program since 2006, http://marinedebris.noaa.gov/, Hawai'i Marine Debris Action Plan, Marine Debris NEWS, NOAA's Marine Debris Blog http://marinedebrisblog.wordpress.com/ |
| Ocean Conservancy | NGO founded in 1972, International Coastal Cleanup since 1986, Movement for Trash free seas http://www.oceanconservancy.org/our-work/international-coastal-cleanup/ |
| Ocean Voyages Institute | NGO founded in 1979, Project Kaisei launched in 2009 http://www.oceanvoyagesinstitute.org/ (San Francisco, California) |
| Operation Clean Sweep | International Program designed to prevent the loss of plastic pellets by plastics industry, http://www.opcleansweep.org/ |
| Plastic Pollution Coalition | Global alliance of individuals, organizations and businesses working to stop plastic pollution, http://plasticpollutioncoalition.org/ |
| Plastiki Expedition | Plastiki Expedition lead by David de Rothchild from San Francisco to Sydney in 2010 http://theplastiki.com/ |
| Project AWARE | International initiative Dive against debris, http://www.projectaware.org/project/dive-against-debris |
| Rozalia Project | Marine debris project launched in 2010 by Rachel Z. Miller, education, cleanups, research http://rozaliaproject.org/ |
| SEA (Sea Education Association) | NGO founded in 1971, expeditions Plastics at Sea, North Pacific Expedition, extensive scientific work http://www.sea.edu/plastics (Falmouth, Massachusetts) |
| Sea Cleaners | Charity founded by Hayden Smith in 2011, cleanups and education, Auckland, New Zealand http://seacleaners.com/ |
| SeaLife Conservation | NGO, education and research, Marine Debris Mapping program (San Francisco) http://sealifeconservation.org/what-we-do/marine-debris/ |
| Seas at Risk | International network of 20 NGOs from 14 EU countries, http://www.seas-at-risk.org/index.php?page=5 |
| SCRIPPS Institution of Oceanography | Scripps Environmental Accumulation of Plastics Expedition SEAPLEX http://seaplexscience.com/ (San Diego, California) |
| Surfrider Foundation | Education Program Rise Above Plastics Speaker Program http://www.riseaboveplasticsprogram.org/about/<br>http://www.surfrider.org/ |
| Sustainable Coastlines Hawaii | NGO formed in 2011, marine debris program, Hawaii, http://sustainablecoastlineshawaii.org/ |
| Sustainable Coastlines New Zealand | Charity founded in 2008, project Love your coast, Pacific Island Research Expedition, marine debris cleanups, education, New Zealand http://sustainablecoastlines.org/ |
| Take 3 | NGO formed in 2009 by Tim Silverwood, A Clean Beach Initiative for raising awareness, http://www.take3.org.au/ |
| Tangaroa Blue | NGO founded in 2004, Australian Marine Debris Initiative, research, Port Douglas, Queensland http://www.tangaroablue.org/ |
| The Ocean Cleanup | Project started in 2012 by Boyan Slat, large-scale cleanup of surface debris in the gyres, http://www.theoceancleanup.com/ |
| Two hands Project | NGO formed in 2010, Initiative for raising awareness, Australia http://www.twohandsproject.org/ |
| UNEP | Global Partnership on Marine Litter – International partnership for collaboration since 2012 http://www.marinelitternetwork.org/home |
| West Coast Governors Alliance on Ocean Health | Alliance founded in 2006 by Governors of California, Oregon and Washington, Marine Debris Action Coordination Team http://www.westcoastoceans.org/marine-debris-action-coordination-team |
| WWF (World Wildlife Fund Australia) | NGO active in Australia in the field of marine debris, Marine Debris Tracking Program, http://www.wwf.org.au/ |

Physical removal of debris from the environment is important for preventing its further impact on biota. Organised volunteer beach cleanups, as well as other marine debris mitigation measures, are a multipurpose tool which may incorporate 3 different activities: research and data collection, removal of debris (which otherwise might be costly), and education and raising awareness (Williams *et al.*, 2013). There are two annual cleanup actions on a global level, the International Coastal Cleanup, organised by Ocean Conservancy, and Cleanup the World (Table XY). These cleanups are done manually, by picking up the rubbish.

Removal of debris from beaches can also be done by mechanical cleaning. Opinions on its impacts on beach organisms, particularly meiofauna, are divided. Gheskiere *et al.* (2006) stated there is no long-term damage to beach communities, as opposed to Gilburn (2012) who claimed that mechanical raking reduces biodiversity of macroinvertebrates of sandy beach ecosystem. Removing debris from the ocean floor could be done by ROV robots (remotely operated vehicles), as was proposed by Miller (2012) and Murphy *et al.* (2012). Furthermore, a young innovator Boyan Slat recently proposed a large-scale cleanup of the surface debris from subtropical gyres (the Ocean Cleanup) (see Table 1.8.1.), which the marine debris community is quite critical about, especially in respect with its feasibility. However, Slat provided a comprehensive feasibility study (download available from http://www.theoceancleanup.com/).

Successful marine debris management requires a detailed understanding of the pollution and of human behavior (Sheavly and Register, 2007), as most litter is generated by the general public. For example, littering is considered to occur only with items which do not have a perceived value (Smith, 1977). Some of the preventative measures that have been used to reduce the generation of waste and encourage sustainable business and activities are marked-based instruments, which include 'taxes, charges, fees, fines, penalties, liability and compensation schemes, subsidies and incentives and tradable permit schemes' (Ten Brink *et al.*, 2009, p. 25), with the following examples:

- **Environmental taxes** – these taxes change prices and consequently the behavior of producers and consumers. For example:
    - tourist tax – taxes charged to coastal tourists for beach cleaning, waste infrastructure and awareness-raising programs
    - Plastic-bag tax and tax on other products
    - landfill tax
- **Environmental charges and fees** cover the cost of goods (e.g. electricity), services (e.g. wastewater treatment) and environmental management and mitigation measures
- **Products fees and charges** are consumer disincentives placed on products with potential negative environmental impacts (e.g. packaging, tyres, batteries)
- **Port reception, ship berthing, and commercial and recreational fishing fees** are paid by boats/vessels upon arrival to a port, since according to MARPOL Annex V all vessels should deliver their wastes
- **Charging for waste services, including landfills**
- **Car parking fees** for parking situated at the waterfront, **beach entrance fees** and **waterfront business charges**
- **Deposit refund systems** for products with negative environmental impacts, such as bottles and cans, where the consumer pays the deposit upon the purchase and receives it back upon return for recycling or disposal
- **Fines, penalties and non-compliance fees** are charged for littering or other type of irresponsible treatment of waste (e.g. illegal disposal of fishing gear, illegal dumpling or so-called 'fly tipping')
- **Liability and compensation schemes** are set up for potential damage, which if occurs, is paid for by the polluter or is drawn from previously established funds

- **Tradable permits** are ‘an economic policy instrument under which rights to discharge pollution or exploit resources can be exchanged through either a free or a controlled permit-market’, however, these permits are not easily applicable for marine debris pollution
- **Incentives and technical or financial support** for debris collection at sea, activities to reduce waste within the coastal community, development of integrated waste management (IWM) systems
- **Green procurement** refers to changing procurement decisions toward more environmentally acceptable products, avoiding for example single-use items which are a common part of marine debris

Along with these market-based instruments, several other options have been used in waste reduction, such as extended producer responsibility (Sachs, 2006), cradle-to-cradle design (Braungart *et al.*, 2007), biodegradable materials (Song *et al.*, 2009; Bilkovic *et al.*, 2012; Johansson *et al.*, 2012), and sustainable packaging design (Svanes *et al.*, 2010). Extended producer responsibility (EPR) refers to product-oriented environmental regulation by which the manufacturers are responsible for their products’ potential impacts after being discarded by the consumers (Sachs, 2006). As opposed to cradle-to-grave, in cradle-to-cradle production design the loop between the start and end of product’s life is closed and all components of the product are recyclable (Braungart *et al.*, 2007).

In addition to green procurement, material flow analysis (MFA) is another control method of quantifying the flow of energy and materials in a defined system. Owens *et al.* (2011) measured household solid waste generation and marine litter accumulation on Kayangel Island, Republic of Palau, from 2006 to 2008. Accumulation of marine debris on the island from ocean-derived sources (0.24 kg capita$^{-1}$ day$^{-1}$) was greater than generation of household waste by the islanders (0.18 kg capita$^{-1}$ day$^{-1}$). Biodegradable household waste was not included in the MFA of the waste stream, due to difficulties estimating the quantities, as most of it is fed to the pigs or composted within the household. Some types of waste were substantially more numerous as stranded marine debris, than generated household waste, such as Styrofoam (515kg and 25 kg, respectively) and PET plastics (1,210kg and 155kg, respectively). The authors noted there was a gradual shift from homegrown to imported packaged food, as it is typical for a developing country, which concomitantly resulted in increased incidence of health issues, such as diabetes.

# 2. Pacific situation – current state of knowledge

## 2.1. Research

Research effort in the Pacific region mainly focuses on the Northern Pacific and coasts of continents surrounding the ocean, while little work has been done in island territories other than Hawaii. Studies on surface, bottom and shore debris are summarised in the Tables within the Chapter 1.6.

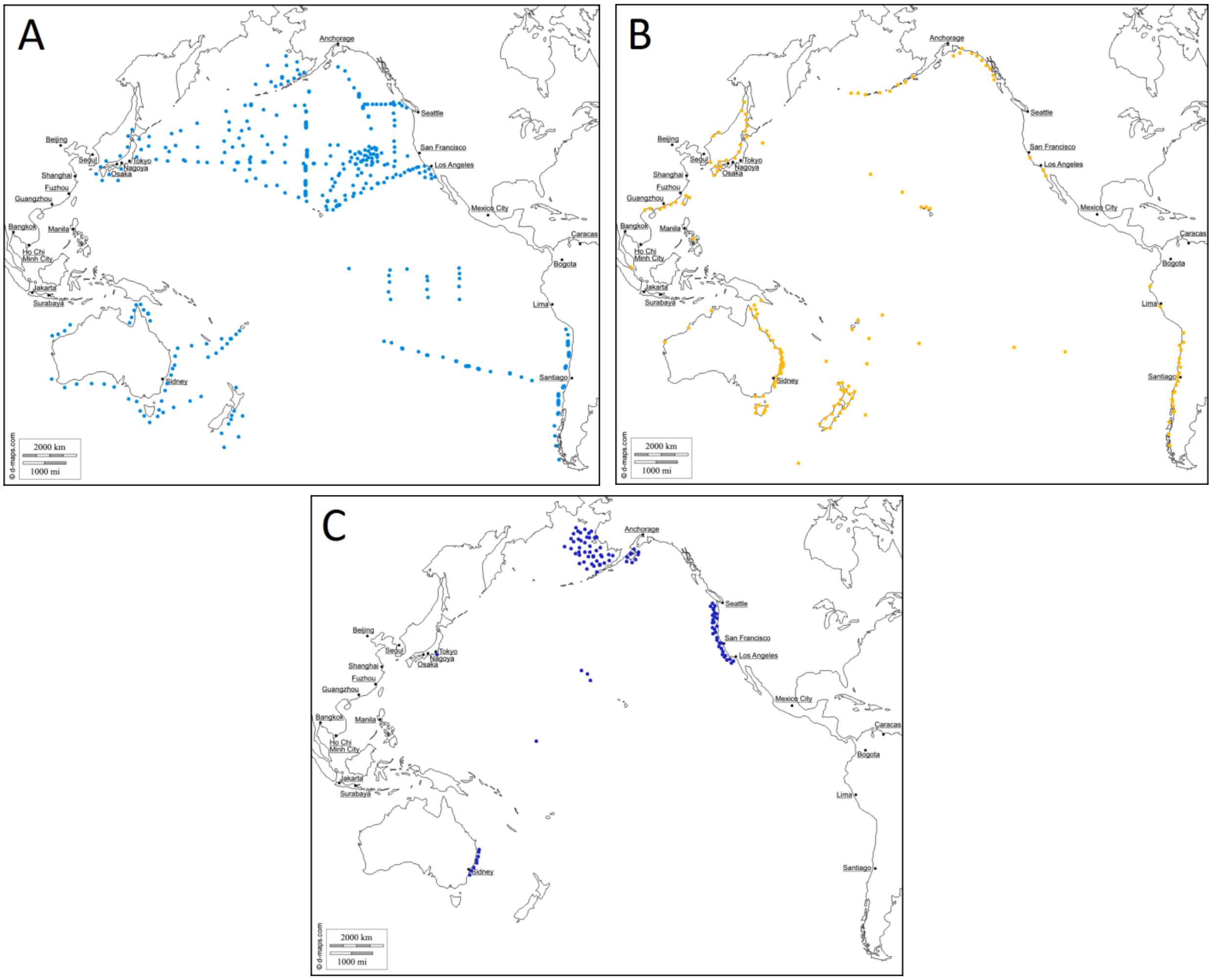

Fig 2.1. Research effort in the Pacific region, roughly depicting survey areas of:
A) surface and near-surface debris, B) shore debris and C) seafloor debris.

Most research effort in the Pacific has been put into surveying floating and stranded debris (Fig. 2.1.). Research on seafloor debris is quite neglected compared to the Atlantic region and the Mediterranean. Furthermore, research on all types of marine debris pollution is virtually absent in Micronesia, Melanesia and Polynesia. Generally, most research has been carried out in developed countries, such as US, Canada, Japan, Australia, New Zealand and Chile.

In the North Pacific, particularly the North Pacific subtropical gyre, marine debris pollution has been extensively studied since the 1970s. Debris densities observed in these waters seem to be greater (Goldstein *et al.*, 2013a) than elsewhere where floating debris is expected to accumulate, such as the recently confirmed South Pacific 'garbage patch' in the Southeast Pacific (Eriksen *et al.*, 2013), as well as the Sargasso sea (Law *et al.*, 2010). The densities of microplastic in the North Pacific Gyre increased from an average of 34,000 pieces km$^{-2}$ in 1972 (Wong *et al.*, 1974) to 334,271 pieces km$^{-2}$ in 1999 (Moore *et al.*, 2001) and 448,000 pieces km$^{-2}$ in 2009 (Goldstein *et al.*, 2013a). Large floating objects increased in occurrence from 4.24 items km$^{-2}$ in 1972 (Venrick *et al.*, 1973) to 1400-3200 items km$^{-2}$ in 2009 (Goldstein *et al.*, 2013a), which is more than 700-fold increase.

The greatest density of stranded debris has been recorded on Japanese beaches (44,521.3 pieces m$^{-2}$), however it should be noted the debris was composed of foamed plastic fragments (Fujieda and Sasaki, 2005). Higesth densities of beach debris are usually found where small fragments or plastic pellets accumulate. Gregory (1978) reported densities of more than 100,000 plastic pellets on 1m of beach front in Auckland in the 1970s. He also found substantial amounts of pellets in some southwest Pacific Islands, such as Tonga (>1000 pellets/m$^{-1}$) and Rarotonga, Cook Islands (>500 pellets/m$^{-1}$). Lee *et al.* (2013) found densities of microplastic on the beaches near Nakdong river estuary, South Korea, in range from 8,205 to 27,606 item m$^{-2}$. Considerable amounts of microplastic were also found on Easter Island shores (805 item m$^{-2}$) (Hidalgo-Ruz and Thiel, 2013), which is situated within the South Pacific subtropical gyre, and on the Hawaiian Islands (>400 pieces m$^{-2}$) within the North Pacific subtropical gyre (McDermid and McMullen, 2004; Cooper and Corcoran, 2010).

As reported by UNEP (2009), research on shore debris in the Northwest of the Pacific region was conducted as a part of the Regional Seas Program, Northwest Pacific Region Environmental Cooperation Centre (NPEC) activities and International Coastal Cleanups. Average densities of marine debris collected during the International Coastal Cleanups in Japan, China, Russian Federation and Korea between 2002 and 2005 were 5.7, 1.2, 0.7 and 0.3 item m$^{-2}$, respectively.

A long-term study (1974-1997) on beach litter on the Kawerua coast, western coast of North Island, New Zealand, demonstrated a significant increase in ocean-borne debris over a 22 year period, particularly hard plastic, foam plastic, plastic sheets and fibre, plastic bottles, containers, lids, buckets, baskets, polystyrene floats, blue strapping loops, nylon lines and twine. In 1974 the density of debris was 0.05 item m$^{-1}$, which increased to 0.14 item m$^{-1}$ in 1997 (Hayward, 1999).

Research on seafloor debris is particularly scarce in the Pacific, however when conducted, recorded densities are exceptionally high in some areas. Richard and Beger (2011) found 773 items of macro debris in 3300m$^2$ of the shallow lagoon of the Majuro Atoll (234,242.4 item km$^{-2}$) and 781 items in 3900m$^2$ on the exposed reef (200,256.4 item km$^{-2}$). Seafloor debris was also found to be abundant in Australian waters of NSW, particularly in Port Stevens (528,000 item km$^{-2}$) (Smith and Edgar, 2014). These densities are generally much higher than elsewhere globally and comparable only to a few locations, such as Curacao and Bonaire in the Caribbean (Nagelkerken *et al.*, 2001; Debrot *et al.*, 2013a,b) and Jordan (Abu-Hilal and Al-Najjar 2009a; Al-Najjar and Al-Shiyab, 2011).

A list of studies on the impacts of marine debris on marine life in the Pacific region is provided in the Chapter 1.7, including the Tables. Ingestion of and entanglement in marine debris by seabirds, sea turtles and marine mammals was studied mostly in the Northeast Pacific (Hawaiian Islands and W coast of US and Canada), the Australian East coast and in New Zealand. Research on the impacts of marine debris is generally deficient in the same areas of the Pacific region where no evaluation of debris abundance, accumulation rates and the potential sources has been undertaken (Fig. 2.1.).

## 2.2. Management

Management of marine debris in the Pacific Region, on a legal basis, is directed by several international treaties covering marine- and land-based sources - MARPOL Annex V, London Convention and Global Programme of Action for the Protection of the Marine Environment from Land-based Activities (GPA) (see Chapter 1.8). The importance of documents addressing marine debris pollution on a global level, the Honolulu Strategy and Honolulu Commitment, has been underlined at the 3rd Intergovernmental meeting of the GPA, where marine debris problem was highlighted as the priority for the period 2012-16.

On a regional level, the marine debris issue in the Pacific Region is addressed by several agreements, initiatives and UNEP's Regional Seas programs (see Chapter 1.8). These include the Noumea Convention for the protection of the South Pacific, Lima Convention (Southeast Pacific), Antigua Convention (Northeast Pacific) and Marine Litter Activity (Northwest Pacific).

*Northwest Pacific*

At the First Intergovernmental Meeting in 1994 (Seoul), China, Japan, Korea and Russian Federation adopted the Action Plan for the Protection, Management, and Development of the Marine and Coastal Environment of the Northwest Pacific (NOWPAP) (UNEP, 2009). Two regional centres have been established in Toyama (Japan) and Busan (Korea) for the implementation of the Action Plan. In 2005, NOWPAP initiated Marine Litter Activity (MALITA) which included cleanup campaigns, a series of marine litter workshops, monitoring guidelines for marine litter on beaches and shoreline and on the seabed and guidelines for improvement of port facilities and services. Additionally, the NOWPAP Regional Action Plan on Marine Litter (RAP MALI) has been developed as the outcome of MALITA (2005-07) and implemented in 2008. Its three key objectives are: prevention of marine litter input to the marine and coastal environment, monitoring of marine litter quantities and distribution, and removal of existing marine litter and its disposal. More details on other work in the Northwest Pacific can be found on http://marine-litter.gpa.unep.org/framework/region-16.htm and http://www.nowpap.org/index.php (NOWPAP).

*Northeast Pacific*

The Regional Seas program in the Northeast Pacific is the most recent one. In 2002, the Convention for Cooperation in the Protection and Sustainable Development of the Marine and Coastal Environment of the North-East Pacific (Antigua Convention) was signed and an Action Plan was developed, covering the issues of sewage and other pollutants, physical alteration and destruction of coastal ecosystems and habitats, overexploitation of fishery resources, and the effects of eutrophication. More information can be found on http://www.unep.org/regionalseas/programmes/nonunep/nepacific/.

*Southeast Pacific*

In 1981 the Convention for the Protection of the Marine Environment and Coastal Area of the Southeast Pacific (Lima Convention) was adopted and came into force in 1986. The convention has 6 protocols, one of which is the Protocol for the Protection of the Southeast Pacific against Pollution from Land-based sources enforced in 1986. The Permanent Commission of the South Pacific (CPPS – Comision Permanente del Pacifico Sur), which acts as the Secretariat of the Convention, is a regional maritime organization for regulation of maritime policies in its member states: Colombia, Chile, Ecuador and Peru. Additionally, Southeast Pacific Action Plan was approved in 1981 and realized through the inter-agency cooperation between CPPS, UNEP, national institutions and other agencies, programmes and Convention Secretariats (UNEP 2009). In 2006 a Memorandum of Understanding to develop activities related with the management of the marine litter in the countries of the region was signed by CPPS and UNEP, which finally resulted in the document 'Regional Programme for the Integrated Management of the Marine

Litter in the Southeast Pacific' published as 'Marine litter in the Southeast Pacific region: a review of the problem' (CPPS, 2007).

*South Pacific*

The development of activities regarding marine debris management of in the South Pacific has been described in the previous Chapter 1.8. SPREP serves as the Secretariat of the Noumea Convention (Convention for the protection of the natural resources and environment of the South Pacific Region). From 2001-2004, SPREP activities were directed by the Action Plan for Managing the Environment of the South Pacific Region, with its marine pollution component Pacific Ocean Pollution Prevention Programme (PAPCOL) which addresses pollution from ships. Furthermore, the PAPCOL Strategy 2010-2014 was developed by SPREP, in partnership with IMO, and approved at the 20th SPREP Meeting in Apia in 2009. Its objectives include (SPREP, 2009, p. 2):

- Prevention of marine pollution through improved awareness, improved planning and operational practices and systems in ports and on vessels and through risk analysis and reduction initiatives.
  - To make environmental considerations a priority in the planning and operations of shipping in the Pacific
  - Conduct on-going risk assessments of vessel types, movements, frequency and cargos to determine high risks locations for marine spills and shipping waste disposal requirements
  - Promote awareness amongst the general public, in particular school age children, on the sources and issues involved in marine pollution and to develop a lifelong respect for the need to protect our coastal and marine environmental heritage
  - Promote International best principles and practices in the maritime business & port development & operation
- Control of marine pollution through the adoption of IMO Conventions and Legislative Framework and educational programs
  - Promote and where possible implement world's best practice in marine environment protection from ship sourced wastes
  - Reinforce the internationally accepted practices of "polluter pays" with the establishment and enforcement of local marine pollution protection legislation and of the "potential polluter pays" with the focus on national levy system to support in-country resources
  - Promote awareness amongst fishing vessel crews, recreational vessel operators, ship owners, shipping agents and others involved in the maritime trade and industry on their legal obligations to comply with local and international rules, legislation and conventions in regards to ship sourced wastes
  - Maintain ongoing activities for capacity building and institutional strengthening in the area of marine pollution prevention and response
- Monitoring of marine pollution through improved surveillance, enforcement, training and new technology developments
  - Extend the knowledge of stakeholders in marine pollution prevention, enforcement and response as it relates to local and international fishing vessels, recreational craft and cruise liners
  - Promulgate uniform policies, consistent principles, guidelines and practices in marine pollution investigation and prosecution
  - Continue to learn the lessons of maritime accidents, incidents and spills that affect the marine environment by sharing case studies and incident reviews from member countries
  - PACPOL must be sufficiently responsive to the pace of change in maritime pollution response, new and emerging technologies, training techniques and expectations
- Mitigation of marine pollution through more effective response planning, incident support, response equipment, systems and training.
  - Take a leading role and pro-active approach to the adoption and implementation of PACPLAN

  - Focus on establishing, maintaining and improving regional spill response support systems for PACPLAN
  - Establish basic policies, practices and procedures to ensure efficient and effective preparedness and response at a national level amongst PICTs
  - Improve the capacity and capability of PICTs to manage and respond to chemical spill incidents
  - Provide direct assistance to PICTs to further develop national and local marine pollution response contingency plans
  - Optimise the use of limited resources and trained spill response personnel in the region through the establishment of a Regional Spill Response Team and systems to ensure fast activation, transfer and deployment of staff and equipment to the incident scene
  - Implement a consistent approach to the establishment of a regional policy and decision making guidelines on “Places of Refuge” for maritime incidents in conjunction with RMP SPC
- Management of marine pollution through the development, funding, implementation and completion of prevention, preparedness and response projects and initiatives
  - Represent SPREP at local, regional and international forums on ship sourced marine pollution prevention and response activities in consultation with PICTs stakeholders
  - Conduct marine pollution prevention projects and initiatives that meet the priorities and needs of PICTs in a coordinated, effective and cost efficient manner
  - Increase the access and delivery of vital and important PACPOL reports, briefs, data and documents via the establishment of a PACPOL web site that is maintained and user friendly
  - Advise promptly appropriate stakeholders including government officials, companies, employees and the public on significant developments in PACPOL
  - Ensure limited financial resources are invested wisely into the high priority projects and activities

In addition to UNEP Regional Seas programs and SPREP activities in the Pacific, US government agencies NOAA and US EPA Region 9 undertake additional action to combat marine debris pollution. The work of both agencies mainly focuses on the North Pacific region. In 2011, U.S. EPA Region 9 developed a Marine Debris Strategy which covers activities such as the ocean assessment and cleanups, and land-based prevention and reduction of marine debris (Fig 2.2.) (U.S. EPA Region 9, 2011). The agency also provided a report on the state of affairs regarding marine debris pollution in the North Pacific, identifying data gaps as well (U.S. EPA Region 9, 2011).

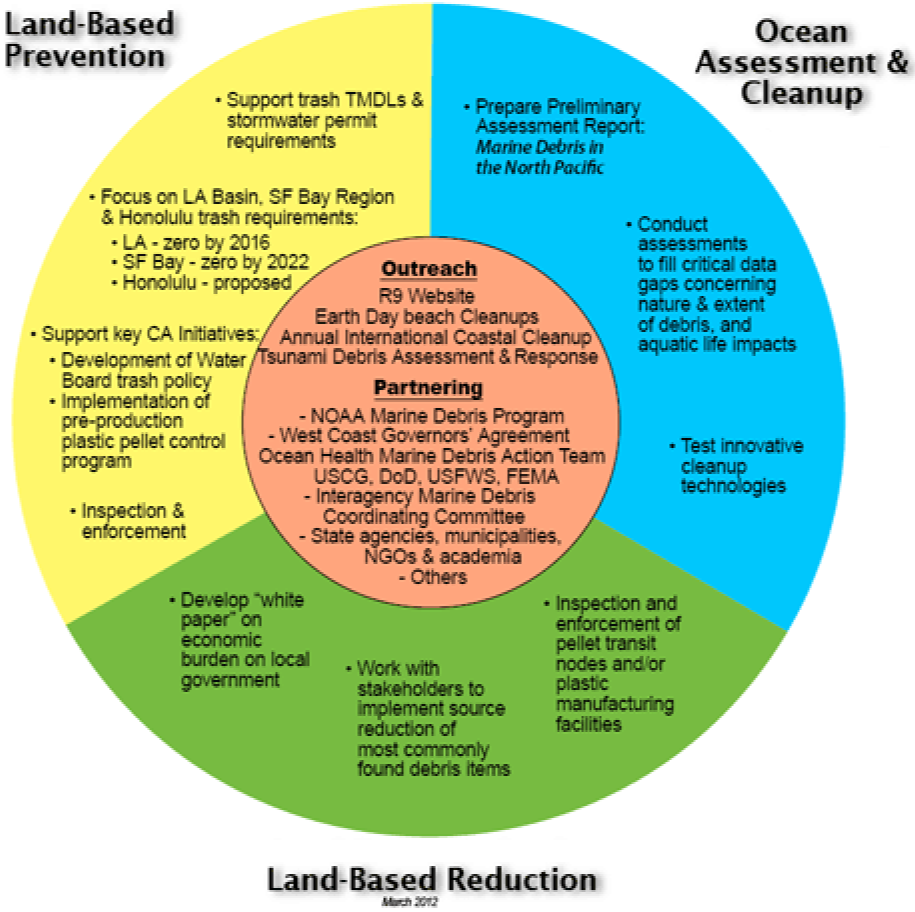

Fig 2.2. Marine Debris Strategy (U.S. EPA, 2014).

NOAA has been particularly active around marine debris issues in the North Pacific, especially the Hawaiian Islands. The agency initiated NOAA Marine Debris Program in 2006 to research, prevent and reduce the impacts of marine debris. Additionally, the Marine Debris Research, Prevention, and Reduction Act (MDRPRA) was signed into US law in December 2006. The accomplishments of Marine Debris Program are provided in a recent progress report (NOAA, 2013). The agency also provides a monthly e-newsletter Marine Debris NEWS.

Furthermore, NOAA developed a Hawaii Marine Debris Action Plan in 2012 with 4 threat reduction goals and strategies to reduce the impacts of marine debris (NOAA, 2012):

- Goal 1: Backlog of marine debris reduced
    - Strategy 1.1: Develop effective methods to locate marine debris accumulations
    - Strategy 1.2: Develop effective systems for reporting marine debris accumulations
    - Strategy 1.3. Develop capacity and coordination mechanisms for marine debris removal
- Goal 2: Incidence of illegal discharge of fishing gear and solid waste at sea decreased
    - Strategy 2.1: Conduct education and outreach to ocean users on marine protection laws, including laws to prevent pollution from ships, ocean dumping laws and proper waste management at sea
    - Strategy 2.2: Provide low-cost and convenient disposal options for gear and solid waste
    - Strategy 2.3: Increase use of fishing materials and practices designed to reduce impacts of marine debris

  - Strategy 2.4: Strengthen marine protection laws, including ship pollution prevention and ocean dumping laws
  - Strategy 2.5: Build capacity to monitor and enforce against illegal discharge of solid waste at sea
- Goal 3: Incidence of abandoned and derelict vessels decreased
  - Strategy 3.1: Educate vessel owners on maintenance and maritime preparedness
  - Strategy 3.2: Develop capacity and coordination mechanisms for abandoned vessel removal and disposal
  - Strategy 3.3: Develop sustainable funding mechanisms and resources for vessel removal and disposal
- Goal 4: Land-based debris in waterways reduced
  - Strategy 4.1: Conduct education and outreach targeted to specific audiences
  - Strategy 4.2: Improve effectiveness of stormwater permits
  - Strategy 4.3: Build capacity to enforce all appropriate laws to reduce discharge of solid waste
  - Strategy 4.4: Employ structural controls and BMPs to prevent land-based debris from entering the ocean
  - Strategy 4.5: Conduct cleanups in watersheds and beaches
  - Strategy 4.6: Decrease used of potential debris items

# 3. Priorities

The state of the marine environment in respect with marine debris pollution in the South Pacific, particularly Melanesia, Micronesia and Polynesia, is largely uninvestigated, considering all aspects of the pollution, including pollution levels, major sources, sinks and pathways, and impacts on marine wildlife and humans. For this reason, general environmental 'scanning' is needed and suggested priorities would include:

- Identification of major sources of marine debris and accumulation points (hot spots). With the assumption that 80% of marine debris originates from land-based activities, research focus should be placed on island states most likely to contribute to marine debris pollution to a greater extent. The factors that should be considered when determining potentially critical sources are:
  - population density and the size of the island land area
  - level of socio-economic development and solid waste management
  - tourism industry
  - susceptibility to natural events (i.e. tsunami, cyclone and king tides)

- Further dispersal pathways of marine debris across all compartments of the marine environment from the potentially significant source points, defined according to the above listed factors, should be investigated. For floating debris ocean current models with different resolutions should be employed. It should be noted that, prior to further use, these proposed source points should be verified in the field (ground-truthed)

- Depending on the results of modeling, identified accumulation hot spots should be ground-truthed. Other reported accumulation points should be examined as well. If accumulations of marine debris, regardless of the source point, occur in areas of greater ecological concern, this should be further investigated for potential impacts of marine debris on marine organisms.

- To determine pollution levels, assessment surveys should be completed minimum for shore marine debris accumulations. Ideally, examination of seafloor, shore and surface marine debris, on both micro- and macro-levels, should be performed to gain a complete insight into the state of the marine environment regarding this pollution, which would facilitate and direct further management steps.

- In respect with negative impacts of marine debris on wildlife, research in the South pacific should focuses on what will be recognised as a priority during the initial observations and evaluations.

Additionally, considering the findings of this review and guidelines provided in the latest international and regional documents on marine debris management (Honolulu Strategy, NOAA Hawaii Marine Debris Action Plan and EPA Marine Debris Strategy), the suggested actions for the Pacific region, in a broader context, include the following:

- *Research and monitoring*

- **An integrated regional approach** to marine debris research encompassing science, legislation, economy and society to be adopted. For example, as marine debris pollution is largely generated by littering and dumping by local people, behavioural studies and public surveys should be incorporated into scientific investigations to allow a better understanding of marine debris sources and the reasons behind destructive human behavior, as well as for subsequent proposal of appropriate management measures. Before and after public surveys can also be used to assess the successfulness of an educational campaign.

- **Development and use of monitoring tools,** with standardised methods, to assess the effectiveness of management measures and legislation, and to measure the progress of marine debris programs and projects. Monitoring should be established within existing national environment departments.

- *Public outreach and education*

- **Public awareness campaigns** on the extent and severity of marine debris pollution and its impacts on the marine environment and humans, engaging stakeholders from government, non-government, academic and private sectors.

- **Systematic environmental education** through the formal schooling system and informally, through targeted religious and community activities. Apart from being related to fishing, shipping and tourism industries, marine debris pollution is, most of all, a 'societal' type of pollution, thus raising awareness on the negative environmental effects of unfavorable human behavior (e.g. littering, dumping, throwaway life-style and overconsumption), on both individual and societal level, plays an important role in tackling this problem.

- **Organised volunteer cleanups**, in addition to being a low-cost solution for removal of debris from the environment and data collection by volunteers and citizen scientists, cleanups are an invaluable educational tool due to first-hand experience.

- **Educating the industries** responsible for contributing to marine debris pollution including the plastics, fishing, shipping and tourism industries.

- *Pollution prevention and reduction*

- **Legislative (regulatory) prevention** - enforcement of existing legislation, development of sector-specific and case-specific policy instruments and implementation of conventions and associated marine debris programs addressing marine- and land-based sources. Gaining control over Illegal discharge of marine debris at sea still seems to be the most challenging task, however building capacity to monitor and police at sea behavior has been suggested, as well as to regularly inspect vessels in ports. Similarly, street littering should be regarded as a significant land-based source of marine debris and addressed accordingly by strengthening and enforcing anti-littering laws. Additionally, illegal waste dumping on land (i.e. fly-tipping), as well as facilitated polluting under the flags of convenience, should be more thoroughly investigated to assess how much these factors contribute to marine debris pollution.

- **Infrastructural prevention -** Infrastructural and organisational improvements of waste and wastewater management are crucial for marine debris pollution prevention. These include:
  - improvement of waste disposal - organised waste collection, moving landfills from the coasts further inland, remediation of wild dump sites,
  - improvement of recycling facilities and facilitation of waste separation through separate bin system,
  - prevention of littering by setting up rubbish bin system in public places,

- installation of waste stands above the ground in areas with ‘wandering’ pets problem (i.e. dogs and cats) to prevent littering
- improvement of storm water system by installation of gross pollutant traps
- deployment of litter booms on main rivers in urban areas

Waste management in islands countries should be more carefully planned and designed considering the limited island land area and frequent natural hazards in the tropical regions, with strong emphasis on waste prevention and reduction.

- **Marked-based instruments and promotion of sustainability.** Paradigm shift from a traditional to sustainable practice in society and economy is an imperative for reduction of waste and marine debris generation. Thus, laying the foundations and building capacity for the development of tools to promote and practice sustainable living should be considered a priority. The implementation of market-based instruments, such as environmental taxes, fees, charges, penalties, non-compliance fees, as well as incentives and financial support, should be encouraged and the instruments used as a regular practice. Items, which are commonly found in marine debris, and are known to cause negative effects in the marine environment, such as disposable plastic bags, soft drink plastic bottles and Styrofoam cups and containers, should be phased-out by banning or placing a fee on them and, if necessary, replaced by a more environmentally acceptable alternative. Other instruments, sustainability measures and strategies, such as corporate sustainability, green procurement, flow material control, extended producer responsibility, cradle-to-cradle and sustainable packaging design, should be incorporated within the existing framework of government and non-government bodies, which should share experience on a regional and global level and follow best practice and successful solutions.

> ‘*Pollution prevention, rather than concentrating on the treatment and disposal of wastes, focuses on the elimination or reduction of undesired byproducts within the production process itself. In the long run, pollution prevention through waste minimization and cleaner production is more cost-effective and environmentally sound than traditional pollution control methods. Pollution prevention techniques apply to any manufacturing process and range from relatively easy operational changes and good housekeeping practices to more extensive changes such as making substitutions for toxic substances, the implementation of clean technology, and the installation of state-of-the-art recovery equipment. Pollution prevention can improve plant efficiency, enhance the quality and quantity of natural resources for production, and make it possible to invest more financial resources in economic development.*’ (Cheremisinoff, 2001).

- *Removal of debris from the environment.*

Accumulated debris in the environment should be regularly removed to prevent further impacts on the marine biota, as well as to avoid generation of microplastics by photo-degradation, mechanical degradation by waves and break down by ingestion.

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# Appendix A

## Tables – Chapter 1.6.

### Table 1.6.1. Research on surface marine debris

### i) Observations: item km$^{-2}$

| Time | Location | Density (item km$^{-2}$) | Method | Reference |
|---|---|---|---|---|
| Pacific and surrounding waters | | | | |
| 1972 | NE Pacific, N of Hawaii | av. **4.24** | ship-based obs., large debris | Venrick *et al.*, 1973 |
| 1985 | NE Pacific, 155W longitude | av. **1.83** (subtropical), **0.94** (subarctic), **0.23** (Bering Sea) | ship-based obs., large debris > 2.5cm | Day and Shaw, 1987 |
| 2000 | NW Pacific, off Japan (around) | av. **<1**<br>(max. 3.31, W coast off Noto Peninsula) | ship-based obs., debris >5cm | Shiomoto and Kameda, 2005 |
| 2002 | SE Pacific, off Chilean coast | **>10** (20-40°S) >20 (near ports),<br><1 (40-50°S) | ship-based obs., large debris | Thiel *et al.*, 2003 |
| 2005 | NE Pacific, N of Hawaii | **<1** (max. around 1.6) | aerial observations, large debris | Pichel *et al.*, 2007 |
| 2009 | NE Pacific ‘Garbage patch’ | av. **1400-3200** | ship-based obs., debris >2cm | Goldstein *et al.*, 2013a |
| 2009-10 | South China Sea, Guandong Province | av. **0.025** (0.02-0.29) | ship-based obs., debris >10cm | Zhou *et al.*, 2011 |
| Atlantic and surrounding waters | | | | |
| 1990 | Sable Island, Nova Scotia | av. **31.6** | ship-based obs., large debris | Dufault and Whitehead, 1994 |
| 1992-94 | Gulf of Mexico | **0.6-2.4** (temporally pooled) | aerial observations, debris > a cup | Lecke-Mitchell and Mullin, 1997 |
| 1997, 2000 | Ligurian Sea, Mediterranean | 1997: **15-25**; 2000: **1.5-3** | ship-based obs., large debris | Aliani *et al.*, 2003 |
| 2002 | Entire longitudinal range from N to S Atlantic | **0-10** (S Atlantic), **0-20** (N Atlantic), **10-100+** (English Channel) | ship-based obs., debris >10cm | Barnes and Milner, 2005 |
| 2006-08 | German Bight, North Sea | **25.1-38.6** (pooled) | ship-based obs., all debris | Thiel *et al.*, 2011 |
| 2013 | South Atlantic ‘garbage patch’ (3-8°E) | **6.2** (the ‘patch’), **1-2.7** (surrounding water), **>100** (S Africa coastal waters) | ship-based obs., debris >1cm | Ryan, 2014 |
| Indian Ocean and surrounding waters | | | | |
| 2012 | Bay of Bengal | av. **8.8** (Bay of Bengal),<br>av. **578** (Straits of Malacca) | ship-based obs., debris >1cm | Ryan, 2013 |

### ii) Collections: a) item km$^{-2}$

| Time | Location | Density (item km$^{-2}$) | Method | Reference |
|---|---|---|---|---|
| Pacific and surrounding waters | | | | |
| 1972 | Across N Pacific, from Tokyo to British Columbia | av. **34,000**<br>(max. around 35N, 143E) | neuston tows, small debris, >0.15mm | Wong *et al.*, 1974 |
| 1984 (?) | Subantarctic Islands and around | NE of Antipodes: **285**; SE of Bounty Is.: | neuston tows, small debris | Gregory, 1987 |

| | | | | |
|---|---|---|---|---|
| | New Zealand | **357**; near Wellington Harbour: **40,909** | | |
| 1985 | NE, 155W longitude | av. **96,000** (subtropical),<br>**3370** (subarctic), **80** (Bering Sea) | neuston tows, small debris, >0.3mm | Day and Shaw, 1987 |
| 1986-89 | Across the North Pacific | av. **316,800** (Japan Sea), greater densities in the W North Pacific | neuston tows, small debris, >0.5mm | Day *et al.*, 1990 |
| 1988-89 | Pacific, South Equatorial current and equatorial countercurrent | South Equatorial current: **137**<br>Equatorial countercurrent: **24** | neuston tows, small debris, > 1mm | Spear *et al.*, 1995 |
| 1999 | North Pacific subtropical high (‘Eastern Garbage patch’) | av. **334, 271** (31,982 to max. 969,777) | neuston tows, small debris, >0.3mm | Moore *et al.*, 2001 |
| 2000-01 | Kuroshio Current area, off Japan (Kii peninsula) | av. **174,000** (max. 3,529,000) | neuston tows, small debris, >0.3mm | Yamashita and Tanimura, 2007 |
| 2008 | Hauraki Gulf, New Zealand | av. **2310 – 16,577** | neuston tows, small debris, >0.25mm | Young and Adams, 2010 |
| 2009 and 2010 | NE Pacific, California to Hawaii (‘Eastern Garbage patch’) | 2009: av. **448,000** (max. 6,553,000 gyre centre); 2010: av. **21,000** (max. 910,000 gyre centre) | neuston tows, small debris, >0.3mm | Goldstein *et al.*, 2013a |
| 2009-10 | South China Sea, Guandong Province | av. **4.92** (0.27-16.8) | trawl net, debris size between 2.5-10cm | Zhou *et al.*, 2011 |
| 2011 | SE Pacific subtropical gyre, from S America to Pitcairn Islands | av. **26,898**<br>(max. 396,341 east of Easter Island, centre of the gyre) | neuston tows, small debris, >0.3mm | Eriksen *et al.*, 2013 |
| 2011-12 | Around Australia and to Fiji | av. **4,256.4** (max. 15,500-23,610.7 at Dampier, Brisbane, Sydney, Hobart, Fiji) | neuston tows, small debris, >0.3mm | Reisser *et al.*, 2013 |
| Atlantic and surrounding waters | | | | |
| 1971 | NW Atlantic, Sargasso Sea | av. **3,500** (max. 12,800) | neuston tows, small debris, >0.3mm | Carpenter and Smith, 1972 |
| 1972 | E US coast, from Cape Cod to the Caribbean | **262, 276** (coast Cape Cod to Florida),<br>**2,707** (Bahamas), **1,292** (Caribbean Is.) | neuston tows, small debris, >0.95mm | Colton *et al.*, 1974 |
| 1977-78 | SE Atlantic, Cape Province, S Africa | av. **3640** | neuston tows, small debris, >0.9mm | Ryan, 1988 |
| 1979 | SE Atlantic, Cape Basin | av. ~**2700** (1,333-3,600) | neuston tows, small debris, >0.3mm | Morris, 1980 |
| 1984 | NW Atlantic | >**11,000** (N Sargasso), **2,500** (S Sargasso),<br>**1,400** (Caribbean), **7,000** (Cape Cod) | neuston tows, small debris, >0.3-0.5mm | Wilber, 1987 |
| 1986-2008 | NW Atlantic, Sargasso Sea | 1986 – av. **15,000**, 1996 – av. **36,000**,<br>2008 – av. **4,000 (**max. 580,000 – in 2007) | neuston tows, small debris, >0.3mm | Law *et al.*, 2010 |
| 1988 | Caribbean Sea, Jamaica | no debris was found in any of the tows | neuston tows, small debris, >1mm | Wade *et al.*, 1991 |
| 1990 | Sable Island, Nova Scotia | av. **84,000** | neuston tows, small debris, >0.3mm | Dufault and Whitehead, 1994 |
| 2010 | NW Mediterranean Sea | av. **116,000** (max. 892,000 Island Elba) | neuston tows, small debris, >0.3mm | Collignon *et al.*, 2012 |
| 2011 | Belgian coastal waters | **Feb:** av. **4,250** (500-13,000)<br>**Jul:** av. **3,500** (1000-9000) | neuston tows, small debris, >1mm | van Cauwenberghe *et al.*, 2013 |
| 2011-12 | Bay of Calvi, Corsica, Mediterranean | av. **62,000** (max. 688,000) | neuston tows, small debris, 0.2-10mm | Collignon *et al.*, 2014 |

### b) item m$^{-3}$

| Time | Location | Concentration (item m$^{-3}$) | Method | Reference |
|---|---|---|---|---|
| Pacific and surrounding waters | | | | |
| 1984, 1994 and 2007 | NE Pacific, Los Angeles coastal waters | 1984 - av. **0.011** (max. 0.82)<br>1994 – av. **0.033** (max. 0.9)<br>2007 – av. **0.016** (max. 3.14) | neuston tows, small debris, >0.5mm | Gilfillan *et al.*, 2009 |
| 1999-2002 | NE Pacific waters | av: **2.23** (1999, E Garbage patch),<br>**0.43** (2000, offshore and gyre),<br>**7.25** (2000, LA coast),<br>**5** (2001, LA coast, max. 18.3),<br>**1.5** (2002, Gyre) | neuston tows, small debris, >0.3mm | Lattin *et al.*, 2004 and Moore *et al.*, 2005 |
| 2006-07 | N Pacific, SE Bering Sea and off W US coast | av. **0.004-0.19,**<br>(max. Bering Sea 0.4, off W coast 3.14) | neuston tows, small debris, >0.5mm | Doyle *et al.*, 2011 |
| Atlantic and surrounding waters | | | | |
| 1972 | NW Atlantic, off New England | av. **0.02-1** (max. 14, Niantic Bay area) | neuston tows, small debris, >0.3mm | Carpenter *et al.*, 1972 |
| 2002-08 | NE Atlantic, Portugal coastal waters | min. **0.002** (off Aveiro),<br>max. **0.036** (off Costa Vicentina) | neuston tows, small debris, >0.18mm | Frias *et al.*, 2014 |
| 2003-04 | Equatorial Atlantic, St. Peter and St. Paul Islands | av. **0.01** | neuston tows, small debris, >0.3mm | Ivar do Sul *et al.*, 2013 |
| 2012-13 | Mediterranean, Sardinia, Gulf of Oristano | av. **0.15**<br>(max. 0.35, offshore, out of the gulf) | neuston tows, small debris, >0.5mm | de Lucia *et al.*, 2014 |
| 2012-13 | Goiana Estuary, Brazil | av. **0.26** | neuston tows, small debris, >0.3mm | Lima *et al.*, 2014 |

Table 1.6.2. Research on subsurface marine debris, occurring in the water column.

| Time | Location | Concentration (item m$^{-3}$) | Method | Reference |
|---|---|---|---|---|
| Pacific and surrounding waters | | | | |
| 2001, 2002 | NE Pacific, Santa Monica Bay, off Los Angeles, NE Pacific gyre | 2001 – av. **3.05** (max. av. **10.8** at 5m depth, after the storm), av. **3.8** (max. av. **6** at 30m depth, before the storm)<br>2002 – av. **1.52** (gyre) | subsurface tows, small debris, >0.3mm, at 5m depth and near-bottom at 30m depth | Lattin *et al.*, 2004 and Moore *et al.*, 2005 |
| 2006, 2007 | NE Pacific, off US west coast | av. **0-0.004** | subsurface tows small debris, >0.5mm, at 212m depth | Doyle *et al.*, 2011 |
| 2006, 2007 | NE Pacific, California current system | 2006 – **0**;<br>2007 – **low** | subsurface tows small debris, >0.5mm, at 210m depth | Gilfillan *et al.*, 2009 |
| 2012 | NE Pacific, off British Columbia to 145°W | av. **2080**<br>(max. **9180** in Queen Charlotte Sound) | subsurface tows, small debris, 0.25 – 5mm, at 4.5m depth | Desforges *et al.*, 2014 |
| Atlantic and surrounding waters | | | | |
| 1960s-1990s | NE Atlantic; 2 routes: Aberdeen – Shetlands and Sule Skerry – Iceland | 1960-70s – **0.01**;<br>1980s – **0.045**;<br>1990s – **0.034** | subsurface tows small debris, >0.28mm, at 10m depth | Thompson *et al.*, 2004 |

Table 1.6.3. Research on marine debris accumulated on the shore: a) shore surface debris (item $m^{-1}$)

| Time | Location | Quantity (item $m^{-1}$) | Proportion of plastic (%) | Debris size | | Reference |
|---|---|---|---|---|---|---|
| | | | | Found | Sampled | |
| Pacific | | | | | | |
| 1972-76 | New Zealand, entire coastline | av. < **50**; max.: Auckland: >>100,000 (Chelsea); Wellington: >40,000 (Petone and Oriental Bay) | only plastic pellets collected | - | pellets | Gregory, 1978 |
| 1972-1982 | N Pacific, Aleutian Islands, Amchitka | 1972: av. **0.16**; 1973: av. **0.22**;<br>1974: av. **0.36**; 1982: av. **0.28** | plastic only collected | - | > 5mm | Merrell, 1984 |
| 1974-97 | New Zealand, North Island, W coast, Kawerua | 1974: **0.05**, 1975: **0.09**; 1981: **0.04**; 1987: **0.08**; 1990: **0.11**; 1993: **0.11**; 1997: **0.14** | In 1997: 48.7 (hard plastic), 36.5 (plastic sheet and fibres), 6.6 (foam), | - | - | Hayward, 1999 |
| 1982-87 | N Pacific, Aleutian Is. | total plastics: 1982 ~ **0.3**; 1987 ~ **0.3** | plastic only collected | - | > 5mm | Johnson, 1990 |
| 1986-89 | Heard and Macquarie Islands | Heard Island annual av.: **0.013**<br>Macquarie Island 6-month av.: **0.009** | 51 -71 (plastic) | - | >1cm | Slip and Burton, 1991 |
| 1989-90 | New Zealand coast, 338 beaches | av. **0.99** | 74.62 (plastic) | - | all debris | Smith and Tooker, 199 |
| 1990 (?) | SW Pacific Islands | max.: Norfolk Is.: **100**;<br>Raoul Is.: **> 50** (N Beach);<br>Fiji – Viti Levu: **>> 100** (Deuba),<br>- Vanua Mbalavu: **24** (East);<br>Tonga: **>>1,000** (Laulea Beach);<br>W Samoa: **20** (Malaeia Beach);<br>Cook Is, Rarotonga: **>>500** (Ngatangiia Harbour) | only plastic pellets collected | - | < 5mm | Gregory, 1990 |
| 1993 | S Pacific, N.Zealand, Chatham Is. | <**0.04** (E shore of Chatham Is.)<br>~ **1** (N and W shores of Chatham Is.) | - | - | large | Gregory, 1999 |
| 2008-10 | Hawaii Islands, Midway Atoll, Sand Island | Jan: av. **16.51**; Apr: av. **7.06**; Aug: av. **2.71** (max. av. 18.45 in November) | 91 (plastic), 7.2 (SF) | - | > 2.5cm | Ribic *et al.*, 2012a |
| Australia | | | | | | |
| 1990 (?) | Australia, Tasmania, Victoria, NSW, Queensland | max.: Tasmania: 0;<br>Victoria: >1,000 Mordialbo;<br>NSW: >> 2,000 (Botany Bay);<br>Queensland: 5 (Bargara and Keppel Sands) | only plastic pellets collected | - | < 5mm | Gregory, 1990 |
| 1990-91 | Tasmania's coastline and WHA | av. **0.3** (World Heritage Area: 0.17) | 65 (plastic) | - | - | Slater, 1991,<br>Slater, 1992 |
| 1992 | Queensland, Deepwater National Park | **0.22-0.45** | majority is plastic | - | all debris | Woodall, 1993 |
| 1996 | Queensland, Cape York, GBR islands | av. **0.11** (max. Moulter Cay: 0.08; Boydong Island 0.16; Quioin Island: 3.63 | 43 (plastic), 11 (PS) | - | >1-2cm | Haynes, 1997 |
| 1996-97 | Northern Territory, Fog Bay, Bare Sand and Quail Islands | 1996: Quail – **0.05**,<br>Bare Sand Island – **0.11**<br>1997: Bare Sand Is. – **0.15** | 45 (all synthetic materials: plastic, rubber, foam) | - | all debris | Whiting, 1998 |
| 2003 | Northern NSW | av. **0.21** | 72-86 (plastic), 1-11 | - | - | Taffs and Cullen, 2005 |

| | | | | | | |
|---|---|---|---|---|---|---|
| | | (Woolie B. 0.14, Minnie Back B. 0.3) | (cig. butts) | | | |
| 2009-10 | Northern NSW | **0.69-16.25** | 49 (plastic), 44 (SF) | - | >5mm | Smith, 2010 |
| Americas | | | | | | |
| North America | | | | | | |
| 1978 | Nova Scotia, Sable Island and Bermuda | Nova Scotia: av. < **10,** Sable Island: ~ **0**<br>Bermuda: av. > **5000** (max. > 10,000) | only plastic pellets collected | - | pellets | Gregory, 1983 |
| Middle America | | | | | | |
| 1988 | Caribbean Sea, Jamaica | **0-77** | 60.2 (plastic), 12 (SF) | Majority 10-15m | > 1cm | Wade *et al.*, 1991 |
| 1991 | East Caribbean Sea, St. Lucia and St Dominica Is. | **4.5-11.2** (St. Lucia),<br>**1.9-6.2** (St. Dominica) | 16 – 51.3 (plastic - St. Dominica – St. Lucia) | - | > 1-2cm | Corbin and Singh, 1993 |
| 1992, 1993 | South Caribbean Sea, Curacao | av. **19-253** (windward side),<br>av. **1-4** (leeward side) | 40-64.2 (leeward – windward) | 23% - 5-10cm;<br>20% > 30cm | >1 cm | Debrot *et al.*, 1999 |
| 1996 | Bahamas, San Salvador Island | av. **0.64** | 47.1 (plastic), 2.6 (SF) | - | - | Alter *et al.*, 1999 |
| 2011 | South Caribbean Sea, Bonaire | windward: av. **291** (9-1640), leeward: av. **1.4** (0.1-5), Lac Bay: av. **~90** (44-116) | 72 (plastic), 16 (SF) | - | > 5cm | Debrot *et al.*, 2013a,<br>Debrot *et al.*, 2013b |
| South America | | | | | | |
| 1994-95 | Brazil, Rio Grande, Cassino Beach | av. **7.4** (0.3-60.7) | 40.3 (plastic) | - | >2cm | Wetzel *et al.*, 2004 |
| 2001-04 | Brazil, Bahia, Costa dos Coqueiros | 2001: av. **0.009**; 2002: av. **0.01**;<br>2003: av. **0.006**; 2004: av. **0.009** | 69.8 (plastic), 0.3 (SF), 0. (foam) | - | - | Santos *et al.*, 2005 |
| 2004-05 | Brazil, Bahia State | winter: av. **3**, summer: av. **6** | 67.25 (plastic),<br>6.5 (PS) | Majority 1-10cm$^2$ | > 1cm | Ivar do Sul *et al.*, 2011 |
| 2005 | Brazil, Bahia, Costa do Dende | av. **9.1** (0.5-30.1) | 76 (plastic), 14 (SF) | - | > 2cm | Santos *et al.*, 2009 |
| Atlantic | | | | | | |
| 1984 | South Atlantic, Prince Edward, Gough, Inaccessible and Tristan da Cunha | Prince Edward: **0.032**,<br>Gough Island: **0.014**<br>Inaccessible Is. and Tristan da Cunha: av. **0.42** | 88 (Prince Edward),<br>84 (Gough Is.),<br>68 (Inaccessible Is. and Tristan da Cunha) | - | > 1cm | Ryan, 1987 |
| 1984, 1987 | South Atlantic, Inaccessible Island | 1984: av. **0.56**<br>1987: av. **0.82** | 62-70 (plastic) | - | > 1cm | Ryan and Watkins, 1988 |
| 1990-95 | Southern Ocean, South Georgia, Bird Island | annual av.: 1990: **0.066**,<br>1992: **0.093**; 1994: **0.064**;<br>1995: **0.21** | 76 (synthetic line),<br>6 (packaging bands),<br>6 (PE bags) | - | - | Walker *et al.*, 1997 |
| 1993-97 | Southern Ocean, Scotia Arc Islands | Adelaide Is: **0**, South Georgia: **0.3**<br>South Sandwich Islands: **0.008-0.29**;<br>South Orkney Is: **0.021**;<br>South Shetland Is: **0.12** | ~ 30-95 (synthetic) | - | - | Convey *et al.*, 2002 |
| 1993-2000 | across Atlantic | 1993 – South Georgia: **0.36**;<br>1994-00 – Scolt Head (UK): **0.49**; | - | - | - | Barnes and Sanderson, 2000 |

| | | | | | | |
|---|---|---|---|---|---|---|
| | | 1996 – Sal (Cape Verde): **1.6**;<br>Sao Miguel (Azores): **0.76** | | | | |
| 1996-2001 | across Atlantic and surrounding waters | 1996 – Rhum Is.(Scotland): **0.1**5;<br>Sao Antao (Cape Verde): **1.36**;<br>1998 – San Blas (Panama): **2.75**;<br>1999 – Sherkin Is. (Ireland): **0.94**;<br>2000 – Jabuka (Croatia): **6.4**;<br>2001 – Tenerife (Canary Is.): **2.24** | - | - | - | Barnes, 2002 |
| 2001-02 | South Atlantic, Falkland Islands | Volunteer Beach, monthly av. **0.08** | 74 (plastic) | - | all debris | Otley and Ingham, 2003 |
| 2001, 2002 | across Atlantic and surrounding waters | Scott Head: **0.63/0.68**; Spitsbergen: **0.04**;<br>Clare (Ireland): **0.28**; Streymoy (Faroe Is.): **0.21,**<br>SW Iceland: **0.23,** Westmann (SE Iceland): **0.17;**<br>Menorca (Baleari): **8.8**;<br>La Gomera ( Canary Is.): **1.91**; Ascension: **3.4**;<br>Dominica (Carribean): **1.5**;<br>Falkland Is.: **0.43**; Shetland: **0.29;**<br>King George **0.18**; Antarctica – Adelaide **0.003**,<br>Trump and Wienke **0**, | 60.2 | - | - | Barnes and Milner, 2005 |
| Europe and Mediterranean | | | | | | |
| 1971, 1974 | UK, Scotland,<br>Loch Scavaig | 1971: **0.9**<br>1974: **4.77** | all plastic | - | - | Scott, 1975 |
| 1983-84 | German Bight, Island Helgoland | annual av. **1.34** | 75 (plastic) | - | - | Vauk and Schrey, 1987 |
| 1989 | Mediterranean | max. Spain: **56**,<br>Sicilia: **1000** | 49-71 (plastic),<br>3.5-9.3 (SF) | - | > 1-2cm | Gabrielides *et al.*, 1991 |
| 1991-93 | UK, England and Wales | 1991: **0.0002**; 1992: **0.0002**;<br>1993: **0.0008** | all medical waste | - | - | Phillip *et al.*, 1994 |
| 2001-10 | Atlantic coast, NW Spain, Galicia | Lanzada beach: 2002: ~ **5**;<br>2006: **5-40**: 2010: **4-54** | 63 (plastic) | - | all debris | Gago *et al.*, 2014 |
| 2005 | Mediterannean, Balearic Islands | av. **36**<br>(min. N Menorca: 17, max. SW Mallorca: 59) | 36-68 (plastic; summer- winter) | - | - | Martinez-Ribes *et al.*, 2007 |
| 2008-09 | Black Sea, SW coast | av. **24**;<br>spring av. **11.65**;<br>autumn av. **45.16**;<br>(max. 197.25 – Ormanli in October 2008) | 62.7 (hard plastic),<br>15.8 (soft plastic),<br>4.4 (synth fibers),<br>4.3 (SF), 3.9 (PU) | 75% <10cm (majority 2-7cm) | > 2cm | Topcu *et al.*, 2013 |
| 2007 | Adriatic Sea, Slovenian coast | av. **12.16** | 64 (plastic), 17 (cig. butts) | - | > 2cm | Palatinus, 2007 |
| 2010, 2011 | Belgian coast | av. **64.3 (3.39 – 217.44)**<br>2010: **59;** 2011: **70** | 95.5 (plastic – 80.9 plastic pellets) | - | micro and macro | van Cauwenberghe *et al.*, 2013 |
| Africa | | | | | | |
| 1984, 1989 | South Africa, Cape Province | 1984 – **491** (micro); **1.09** (macro)<br>1989 – **678** (micro); **2.99** (macro) | 90 | - | < 2 cm and<br>> 2cm | Ryan and Moloney, 1990 |
| 1994-95 | South Africa, SE coast | av. **37.4** (9.3 -147.2) | 83.4 (plastic), 8.5 | 33.3% < 1cm$^2$, | > 1-2cm | Madzena and Lasiak, |

| | | | | | | |
|---|---|---|---|---|---|---|
| | | | (SF) | 23% 1-10cm$^2$ | | 1997 |
| Indian Ocean | | | | | | |
| 1996-2002 | Indian Ocean | accumulated in 1 year:<br>Sri Lanka: **2.77**; Maldives: **2.6**;<br>N. Tanzania: **2**; Diego Garcia: **1.6**;<br>Christmas Is.: **30**; Cocos Is.: **6.87**;<br>N Mozambique: **2.12**; S Mozambique: **1.01**;<br>Rodrigues Is. (Mauritius): **4.41**;<br>SW Madagascar: **1.1** | - | - | > 1cm | Barnes, 2004 |
| Asia | | | | | | |
| 1984, 1995 | Indonesia, Thousand Islands Archipelago | 1984: av. **0.1 – 10.5**;<br>1995: av. **0.3 – 29.1** | 93 in 1985, 80 in 1995 (PS blocks, pl. bags and footwear) | - | visible | Willoughby, 1997 |
| 1988-89 | Israeli coastline | May 1988: av. **8.06**; May 1989: av. **11.97** (max. 17.08 in June 1988 on Akhziv Beach) | 70.9 (plastic), 3.5 (SF) | majority 2-30cm | > 2cm | Golik and Gertner, 1992 |
| 1989 | Israel, Cyprus | max. Cyprus: **34**,<br>Israel: **12** | 49-71 (plastic),<br>3.5-9.3 (SF) | - | > 1-2cm | Gabrielides *et al.*, 1991 |
| 1990-91 | Israel, entire coastline | min. **0.8** (Mikhmoret), max. **34** (Palmahim Beach) | 70 (plastic) | - | visible | Bowman *et al.*, 1998 |
| 2002 | Omani coast, Gulf of Oman | av. **1.78**<br>(min.: 0.43 – As-Seeb,<br>max.: 6.01 – Barka) | 61 (plastic), 18.5 (cig. butts – as organic),<br>3 (SF) | - | all debris | Claereboudt, 2004 |
| 2011 (?) | Saudi Arabia, Obhur Coastline, Jeddah | **1.72-12.54** | ~ 16 plastic<br>(food waste included in study: ~55%) | - | > 2cm | Kitto *et al.*, 2011 |

## b) shore surface debris (item m$^{-2}$)

| Time | Location | Quantity (item m$^{-2}$) | Proportion of plastic (%) | Debris size | | Reference |
|---|---|---|---|---|---|---|
| | | | | Found | Sampled | |
| Pacific | | | | | | |
| 1991 | S Pacific, Pitcairn Islands | Ducie Atoll: **0.12,** Oeno Atoll: **0.35** | ~ 65 (plastic) | - | - | Benton, 1995 |
| 2011-12 | New Zealand, North Island, Ohinau Island | 0.031 (seabirds colony) | plastic only collected | - | - | Buxton *et al.*, 2013 |
| 2014 (?) | Chilean coast, near Elqui, Maipo, Maule and Biobio rivers | max. at Maipo beach:<br>plastic density – **3.4**<br>PS density – **2.1** | 51-60 (plastic),<br>4-33 (PS),<br>3-19 (cig. butts) | - | > 1.5cm | Rech *et al.*, (In press) |
| Australia | | | | | | |
| 1994 | Northern NSW | av. **0.00001** | 39.4 (plastic) | - | all debris | Frost and Cullen, 1997 |
| 1996 | NSW, entire coast | av. **0.056**;<br>0.006 – Newtons Bay<br>0.17 –Newport Bay and Sydney | 73 (plastic), 13 (recreational and commercial fishing) | - | >5mm | Herfort, 1997 |

| | | | | | | |
|---|---|---|---|---|---|---|
| 2000 | NSW, Greater Sydney Area | av. **0.13**<br>(0.01 MacMasters; 0.58 North Era) | 89.8 (plastic) | - | - | Cunningham and Wilson, 2003 |
| 2002 | WA, Broome, Cable Beach | av. **0.25** (0.02-0.53) | av. 61.4 (cig. butts), 12.3 (plastic), | - | - | Foster-Smith t al., 2007 |
| 2011 | NSW, Coffs Harbour | **av. 0.24** | 91.4 (plastic) | - | >5mm | Smith and Markic, 2013 |
| Americas | | | | | | |
| North America | | | | | | |
| 1987-88 | Gulf of Mexico, Louisiana coast | av. **012 – 0.19** | 47 (plastic), 16 (PS) | - | - | Lindstedt and Holmes, 1989 |
| 1994-95 | US, New Jersey, Cliffwood Beach | **2.7-3.7** | 42.5 (plastic), 17.7 (SF), | 51% < 4cm | - | Thornton and Jackson, 1998 |
| 2009-10 | Californian coast, Monterey Bay | av. **1** (0.03-17.1) | 41 (SF), 36 (plastic), cig. butts (6) | majority 2-20mm (av. Size 12.4mm) | <2mm,<br>2-20mm,<br>>20mm | Rosevelt *et al.*, 2013 |
| Middle America | | | | | | |
| 1987 | Honduras, Mosquitia Coast | **50-200 pellets** | 91.5 (polymer origin) | - | - | Cruz *et al.*, 1990 |
| 1990-91 | Panama, Caribbean coast | av. **3.6** | 56 (plastic), 31 (SF) | - | visible | Garrity and Levings, 1993 |
| 2000 | Pacific coast, Baja California, Ensenada | av. **1.53** | 16.3 (plastic) - note: natural debris incl. in the study (>50%) | - | - | Silva-Iniguez and Fischer, 2003 |
| 2011 | South Caribbean Sea, Bonaire | Lac Bay: av. **11.77** (5.8-23.2) | 72 (plastic), 16 (SF) | - | > 5cm | Debrot *et al.*, 2013a,<br>Debrot *et al.*, 2013b |
| South America | | | | | | |
| 2001 | Brazil, Armacao dos Buzios | av. **0.14** (max. 0.23 – Azeda Beach) | 44 (cig. butts), 36.8 (plastic) | - | visible | Oigman-Pszczol and Creed, 2007 |
| 2001 | Brazil, Pernambucco | av. **0.2 – 0.39** | 89.4 (plastic) | - | - | Araujo and Costa, 2004 |
| 2002 | Brazil, Varzea do Una | dry season: av. **11.73**, rainy season: av. **17.67** | plastic only collected | - | - | Araujo and Costa, 2007 |
| 2003-04 | Brazil, Santa Catarina Island | av. **1** (0.13-4.99) | ~90 (plastic) | - | visible | Widmer and Hennemann, 2010 |
| 2006 | Brazil, Sao Vicente Estuary | av. **1.33** | 37.28 (plastic), 1.37 (SF), 0.42 (foam) | - | - | Cordeiro and Costa, 2010 |
| 2008 | Chile, entire coast | av. **1.8** (max > 20 Antofagasta beach) | majority was plastic | - | all debris | Bravo *et al.*, 2009 |
| 2009 | Peru, Punta Veleros and Lima | av. **3.9** | majority was plastic | - | all debris | Thiel *et al.*, 2011 |
| 2014? | Brazil, Salvador | **0.3** (Aleluia) – **2** (Ondina) | 87 (plastic) | - | > 1cm | Leite *et al.*, In Press |
| Atlantic | | | | | | |
| 1993-97 | Southern Ocean, S. Shetland Islands, Livingston Is. | 1993/94: **0.65**; 1994/95: **1.02**: 1995/96: **1.52**; 1996/97: **0.46** | 94.3 (plastic) | - | - | Torres *et al.*, 1997 |

|  |  |  |  |  |  |  |
|---|---|---|---|---|---|---|
| Europe and Mediterranean |  |  |  |  |  |  |
| 1984, 1994 | UK, Edinburgh, Cramond Beach | 1984: av. **0.35**<br>1994: av. **0.8** | - | - | visible | Velander and Mocogni, 1998 |
| 1993 | Ireland, Inch Strand | **0.22** | ~ 68 (plastic) | - | - | Benton, 1995 |
| 2001-03 | UK, Scotland, Firth of Forth | average for 2.5 years: **0.99**<br>(max. 3.06 Dalgety Bay) | 47 (plastic), 5.3 (PS) | - | - | Storrier *et al.*, 2007 |
| 2004 | Turkey, Gulf of Fethiye, Fethiye Beach | **1.03** | 47.6 (plastic) | - | visible | Triessnig *et al.*, 2012 |
| 2007 | Adriatic Sea, Slovenian coast | av. **2.8** | 64 (plastic), 17 (cig. butts) | - | > 2cm | Palatinus, 2007 , |
| 2008-09 | Black Sea, SW coast | av. **0.88**;<br>spring av. **0.41**,<br>autumn av. **1.57**;<br>(max. 5.06 – Ormanli in October 2008) | 62.7 (hard plastic),<br>15.8 (soft plastic),<br>4.4 (synth fibers),<br>4.3 (SF), 3.9 (PU) | 75% <10cm (majority 2-7cm) | > 2cm | Topcu *et al.*, 2013 |
| 2011 (?) | Malta | av. **50.3**<br>(max. 167 – Ghajn Tuffieha Beach) | only plastic pellets collected | av. 3.4mm | pellets | Turner and Holmes, 2011 |
| Asia |  |  |  |  |  |  |
| 1990-91 | Israel, entire coastline | from **0.03-0.05** to **0.13-0.32**<br>(max. **0.88** – Givat Olga Beach) | 70 (plastic) | - | visible | Bowman *et al.*, 1998 |
| 1991-92 | United Arab Emirates, entire coastline | av. **1.19** | 27.1 (plastic fragments), 7.1 (pl. bags), 8.5 (PS blocks) | - | all debris | Khordagui and Abu-Hilal, 1994 |
| 1994 | Indonesia, Thousand Islands Archipelago | av. **1-54**<br>(Islands Hantu Besar – Bidadari) | - | - | - | Uneputty and Envans, 1997 |
| 1994-95 | Jordan, Gulf of Aqaba | annual av.: 1994 – **3**; 1995 – **5**<br>(max. 7.38 at Central Power Station in 1994) | ~ 50 (plastic), ~ 8 (9tyrofoam) | - | >2cm | Abu-Hilal and Al-Najjar, 2004 |
| 1994-95 | Jordan, Gulf of Aqaba | 1994: av. **38,078**; 1995: av. **43,947**<br>(max. 1994: 89,754, 1995: 81,077 Marine Nature Reserve) | only plastic pellets collected | mostly 2-3mm pellets | pellets | Abu-Hilal and Al-Najjar, 2009b |
| 2000 | Japan and E Russia | av. **3.41** (Japan), av. **0.2** (Russian coast) Ɛ | 72.9 (plastic), 19.3 (SF) | - | - | Kusui and Noda, 2003 Ɛ |
| 2005 | Japan, Hiroshima Bay | av. **44,521.3** | 99.5 (SF), 0.4 other plastic | 98.5% <10mm | >2mm | Fujieda and Sasaki, 2005 |
| 2009 | Papua New Guinea, Motupore Island | av. **15.3** (max. 78.3) | 89.7 (plastic) | - | > 5cm | Smith, 2012 |
| 2009-10 | South China Sea, Guandong Province | av. **0.000033** (0.000003 – 0.00038) | 41.4 (plastic), 5.3 (SF) | - | - | Zhou *et al.*, 2011 |
| 2009-11 | Taiwan, Cijin Island | av. **0.9** | 78.3 (plastic incl. rubber) | - | - | Liu *et al.*, 2013 |
| 2010 | Malaysia, Port Dickson | av. **0.26-0.5** (Teluk Kemang – Pasir Panjang) | 64 (Teluk Kemang),<br>46 (Pasir Panjang) | - | - | Khairunnisa *et al.*, 2012 |
| 2011 (?) | Saudi Arabia, Obhur Coastline, Jeddah | av. **8.4** | ~ 16 plastic (Food waste included in study: ~55%) | - | > 2cm | Kitto *et al.*, 2011 |

| | | | | | | |
|---|---|---|---|---|---|---|
| 2011-12 | India, Mumbai | av. **11.6** (0.25-282.5) | plastic only collected | >80% - 5-100mm | all debris | Jayasiri *et al.*, 2013 |
| 2012 | South Korea, Nakdong River Estuary | before and after rain: micro: **8,205** and **27,606** meso: **238** and **237**, macro: **0.97** and **1.03** | plastic only collected | - | > 1mm | Lee *et al.*, 2013 |
| 2012-13 | N Taiwan | av. **0.02-0.61** | 71.84 (plastic), 8.17 (PS), 5.44 (nylon) | - | - | Kuo and Huang, (In press) |

## c) shore sediment debris (various units)

| Time | Location | Quantity | Proportion of plastic (%) | Debris size distribution | Method | Reference |
|---|---|---|---|---|---|---|
| Pacific and surrounding waters | | | | | | |
| 1998-2000 | Japan, Kagoshima prefecture | **290.5** pieces per L of sediment – Kagoshima Bay | 92.6 (SF),<br>1.1 (plastic) | 91% 0.3-4mm | debris > 0.3mm | Fujieda *et al.*, 2002 |
| 2000 | Japan and Russia | item m$^{-2}$: ɛ g m$^{-2}$:<br>Russia **31.3** Russia **8.8**,<br>Japan **2610** Japan **13.6** | 87.1 (SF),<br>10.6 (plastic),<br>1.8 (resin pellets) | - | top 5cm of sand, separated by flotation and sieved through 0.3mm mesh, debris >0.3mm | Kusui and Noda, 2003 ɛ |
| 2001-03 | Hawaiian Islands, Midway Atoll, French Frigate Sh., O'ahu, Molok'I, Hawai'i | av. **23.38** g per 20L of sed. (max. **441.1** item L$^{-1}$ – Cargo beach, 12.68 item L$^{-1}$ Turtle Beach – Midway Atoll) | 72 (plastic by weight), plant mat. Included | majority 2.7-4.75mm,<br>43% 1-2.8mm | top 5.5cm of sand, sieved through 1, 2.8 and 4.75mm mesh, debris > 1mm | McDermid and McMullen, 2004 |
| 2008 | Hawaiian Islands, Kauai | **484** pieces per 5m$^{2}$ per day – Maha'ulepu Beach, Kauai | only plastic included | majority < 1cm$^{2}$ | top 3cm of sand from 5m$^{2}$, collected daily for 11 days | Cooper and Corcoran, 2010 |
| 2011 | S Pacific, Chilean coast and Easter Island | Chilean coast av. **27** item m$^{-2}$ (max. in Chile: Aysen 169 item m$^{-2}$), Easter Island: **805** item m$^{-2}$ | only plastic included | 85% 1 – 4.75mm<br>15% 4.75 – 10mm | top 2cm of sand, sieved through 1mm mesh,<br>debris >1mm | Hidalgo-Ruz and Thiel, 2013 |
| Atlantic and surrounding waters | | | | | | |
| 2004 (?) | UK coastline, sandy beaches | **0.5** fibers per 50 mL of sed. | - | - | density separation (saturated solution of NaCl) | Thompson *et al.*, 2004 |
| 2008 | Atlantic, Fernando de Noronha Archipelago | g of debris per g of sediment:<br>Windward beaches: **4.6 x 10$^{-3}$**,<br>Leeward beaches: **0.3 x 10$^{-3}$** | 65 (plastic),<br>23 (pl. pellets),<br>5 (fishing line),<br>0.5 (PS beads) | 81% <20mm,<br>19% >20mm | top 2cm of sand sieved through 1mm mesh,<br>debris >1mm | Ivar do Sul *et al.*, 2009 |
| 2010 (?) | Brazil, Recife, Boa Viagem Beach | av. **0.29** item cm$^{-2}$ | only microplastic included | 64% 1-20mm, | top 2cm of sand sieved through 0.5-1mm mesh,<br>debris < 2cm | Costa *et al.*, 2010 |
| 2010 | Portuguese coastline | av. **185.1** item m$^{-2}$ | only plastic included | 27% 4mm, 18% 5mm, 14% 3mm, 10% >10mm | top 2cm of sand sieved through 2.5x3.5mm mesh, density separation (NaCl solution),<br>debris < 2cm | Martins and Sobral, 2011 |
| 2010-11 | Belgian coast, sandy beaches | av. **17** particles L$^{-1}$ of sand,<br>or **13** particles kg$^{-1}$ of dry sand | only microplastic | - | top 5cm of sand, elutriation and NaI extraction + sieving through 1 and | van Cauwenberghe *et* |

| | | | | | | |
|---|---|---|---|---|---|---|
| | | (7.2 -20.4 particles kg$^{-1}$) | included | | 0.035mm mesh,<br>debris 0.035 – 1mm | *al.*, 2013 |
| 2011 | Germany, E Frisian Islands | Kacheloplate: **0-621** granules per 10g sed., **1-14** fibres per 10g<br>Spiekeroog: **15-48** (av. **38**) per particles 10g sed | only microplastic included | great majority < 0.1mm | top 1cm of sand, separated by flotation (ZnCl2 solution), sieved through 0.0012mm mesh, debris >0.0012mm | Liebezeit and Dubaish, 2012 |
| 2011 | Germany, E Frisian Islands, Norderney Island | fibres: **4-213** fibres kg$^{-1}$,<br>< 1mm: av. 1-2 particles kg$^{-1}$ | only plastic included | - | top 3cm of sand, sieved 1mm mesh, air-induced overflow extraction and density separation (NaI solution) | Dekiff *et al.*, 2014 |
| 2012 | Portugal, entire coastline | av. **2421** item m$^{-2}$ of beach surface | 98 (plastic – 53% resin pellets) | - | top 2cm of sand sieved through ~3mm metal mesh | Antunes *et al.*, 2013 |
| 2012 | Canada, Nova Scotia, Eastern Passage | ~ 20 to ~ 80 microplastics per 10g of sediment | only microplastic included | - | top 3-4cm of sand, flotation separation (NaCl solution), sieved through 0.0008mm mesh | Mathalon and Hill, 2014 |
| 2013 | Atlantic, Canary Islands | Lanzarote: max. **109** g L$^{-1}$<br>La Graciosa: **90** g L$^{-1}$<br>Fuerteventura: **30** g L$^{-1}$ | only microplastic included | - | top 1cm, density separation in salt water, debris <5mm | Baztan *et al.*, (In press) |
| 2014 (?) | Brazil, Sao Paolo, | **2.56** (Felix B.) – **5,385.07** (Santos B.)<br>Santos: at 0.1m – **5,385.07** (item m$^{-3}$) up to 1m – **23052.76**; 2m – **29775.79** | Only pellets included | - | up to 2m deep, 0.1m intervals, density separation (NaCl), sieved through 1mm mesh | Turra *et al.*, 2014 |
| Indian Ocean and surrounding waters | | | | | | |
| 2004 | India, Alang-Sosiya ship breaking yard | av. **81.43** mg kg$^{-1}$ of sediment | only plastic included | - | top 5cm of sand, filtered through Whatman GF/A | Reddy *et al.*, 2006 |
| 2005 | W Indian coast, Karnataka | av. **121.4** vs. **78** g m$^{-2}$ (hind vs. mid dunes), **13.3** vs. **0.15** item m$^{-2}$ (mid vs. hind dunes) | only plastic included | - | surface and depth collection, up to 30cm | Sridhar *et al.*, 2009 |
| Worldwide | | | | | | |
| 2004-07 | Australia, Oman, UAE, Chile, Philippines, Portugal, Azores, USA, South Africa, Mozambique, UK | pieces per 250mL of sediment:<br>**1-10** – California, E Australia, Philippines (Malapascua Island)<br>**11-20** – Chile, UAE, Oman<br>**21-30** – Japan, S Africa, Mozambique, W Australia<br>**31-40** – Azores, Portugal, UK | only microplastic included | UK: 65% <1mm | top 1cm of sediment,<br>density separation (saturated solution of NaCl) | Browne *et al.*, 2010, Browne *et al.*, 2011 |

Table 1.6.4. Research on marine debris present on the sea floor: a) sea floor surface debris (item km$^{-2}$)

| Time | Location | Quantities (item km$^{-2}$) | Proportion of plastic (%) | Debris size distribution | Method | Reference |
|---|---|---|---|---|---|---|
| Pacific and surrounding waters | | | | | | |
| 1988 | N Pacific | off Oregon: **149.6,** E Bering Sea: **7.5**<br>Norton Sound: **1.9** | 12-51 (plastic) | - | bottom trawls, 3.2-12.7cm mesh, debris > 3.2cm, depth 100-675m | June, 1990 |
| 1993-94, 1997-98, 2002, 2007 | US West Coast, central and southern California | Southern California: **320**;<br>Central California: **6900** | 40-98 | - | 2-person submersible Delta max. depth 365m | Watters *et al.*, 2010 |
| 1994-95 | Alaska, Kodiak Island, Chiniak Bay | **42** | Only crab pots studied | - | side-scan sonar, 2-person submersible and ROV, 100-150m depth | Stevens *et al.*, 2000 |
| 1994-1996 | N Pacific, Alaskan Peninsula and Kodiak Island | **11.2-147.4**<br>(max. Chiniak Bay on Kodiak Is.) | ~ 45-60 (plastic),<br>~0.5-3 (SF) | - | bottom trawls, 400-mesh (?) | Hess *et al.*, 1999 |
| 1995-2000 | Japan, Tokyo Bay | 1995: **338**;<br>2000: **185** | - | - | - | Kuriyama *et al.*, 2003 |
| 1999-2001 | NW Hawaiian Islands, Lisianski Is. (LI), Pearl and Hermes Atoll (PHA), Kure Atoll (KA) | 1999: LI **62.2**, PHA **27.3**<br>2000: LI **17**, PHA **41**, KA **165**<br>2001: LI **16**, PHA **47**, KA **116** | 86-91 (trawl netting) | 74-88% <5m$^2$ | visual census by towed snorkels, depth ~10m | Donohue *et al.*, 2001, Boland and Donohue, 2003 |
| 2005-2010 | Australia, northern NSW | max. **1,400,000** (Pipeline, Port Stevens) and **124,000** (Coffs Harbour)**,**<br>SW Rocks: **20,000-28,000**<br>SIMP: **18,000-20,000,** | 33 (plastic, mostly fishing line) | - | visual census by SCUBA divers | Smith, 2010, Smith and Edgar, 2014 |
| 2007-08 | US West Coast, from Washington to US-Mexico border | av. **67.1** (2007-2008), at diff. depths:<br>55-183m: **25 – 35**<br>184-549m: **65 – 55**<br>550-1280m: **100 – 160** | - | - | bottom trawls, Aberdeen net, 38-140mm mesh, debris > 3.8cm, depth 55-1280m | Keller *et al.*, 2010 |
| 2009-10 | South China Sea, Guandong Province | av. 0.27 – 2.16 (max. 2.16 at Jieyang, trawl survey)<br>av. 3.28 (at Sanya, dive survey) | 40 (plastics) | - | Bottom trawls, dive surveys, max. depth 9m | Zhou *et al.*, 2011 |
| 2011 (?) | W Pacific, Marshall Islands, Majuro Lagoon | av. **140,536** ɛ<br>min. **5,000** (exposed reef)<br>**200,256 – 234,242** (lagoon) | 8 (soft plastics) | - | visual census SCUBA and snorkelers, debris >2cm, max. depth 10m | Richards and Beger, 2011 ɛ |
| Atlantic and surrounding waters | | | | | | |
| 1992-93 | Atlantic, French cont. shelf, Bay of Biscay and Seine Bay | min. – Aug 1993: **26.3**<br>max. – Feb 1993: **494**<br>(av. Biscay bay – 203) | 80 (plastic) | - | Vendean bottom trawl, 55mm mesh, debris > 5.5cm, max. depth 100m | Galgani *et al.*, 1995a |
| 1992-98 | European coastal waters | Baltic Sea: **126**, North Sea: **156**; | av. 51.8 | - | bottom trawls: 10-20mm mesh, | Galgani *et al.*, |

| | | | | | | |
|---|---|---|---|---|---|---|
| | | Channel East: **17.6**;<br>Celtic Sea: **528**; Gulf of Lion: **143**;<br>Bay of Seine: **72**;<br>Bay of Biscay: **142**;<br>NW Mediterranean: **1,935**;<br>East Corsica: **229**;<br>Adriatic Sea: **378** | (plastic) | | debris >1-2cm,<br>+ observations by IFREMER manned submersibles Cyana and Nautile,<br>Depth 50 – 2700m | 2000 |
| 1993-94 | Mediterranean, Gulf of Lion | max. Marseille: **33,237**<br>Gulf of Lion: **225 – 991**<br>Corsica: **743 – 1,487** | 77.1 (plastic) | - | bottom trawls (Drezden and Boc73): 10, 55 and 100 mm mesh, debris >1cm | Galgani *et al.*, 1995b |
| 1994-95 | Caribbean, Curacao | public beach: av. **365,000**;<br>private beach: av. **10,000**<br>(max. 660,000 – Groot Knip) | 47 (plastic) | - | SCUBA/snorkel (?) | Nagelkerken *et al.*, 2001 |
| 1996-2001 | Uruguay-Argentina border, Rio de la Plata Estuary | av. < **100**<br>(max. 360) | Majority plastic and plastic bags | - | bottom trawl, 102-200mm mesh, debris > 10cm | Acha *et al.*, 2003 |
| 1997, 1998 | W Greece gulfs | Patras Gulf: **240**,<br>Echinadhes Gulf: **89** | 79-83 plastic),<br>2 (nylon) | - | bottom trawl, 15mm mesh,<br>max. depth 360m | Stefatos *et al.*, 1999 |
| 1999-2011 | N Atlantic and European Seas | Atlantic: **970** (Norvegian cont. shelf),<br>max. submar. canyon **6620** (Lisbon),<br>seamount **1460** (Condor seamount),<br>Mediterranean: **40** (Gulf of Lion),<br>max. submar. canyon **3210** (Blanes) | 41 (plastic) | - | imaging technology (mainly in Atlantic) and fishing trawls (mainly in the Mediterranean),<br>20-40mm mesh size,<br>debris < 2cm | Pham *et al.*, 2014 |
| 2000 | US, Florida Keys National Marine Sanctuary | av. **13,681.6** | - | - | SCUBA/snorkel (?)<br>max. depth 12m | Chiappone *et al.*, 2002 |
| 2000-03 | W and S Greece,<br>4 gulfs | av. **165.9**;<br>Patras Gulf: E 188, W 437;<br>W Gulf of Corinth: 116;<br>Echinadhes Gulf: 72<br>N Lakonikos Gulf: 85 | 56 (plastic) | - | bottom trawls, 15mm mesh,<br>debris > 1.5cm,<br>depth 15-320m | Koutsodendris *et al.*, 2008 |
| 2001 | Brazil, Rio de Janeiro, Armacao dos Buzios | av. **29,000** | 64 (plastic) | - | visual census, snorkelers, shallow | Oigman-Pszczol and Creed, 2007 |
| 2002-2011 | Arctic, Fram Strait, Hausgarten Observatory | 2002: **3,635,** 2004: **1,214**,<br>2007: **7,28,** 2011: **7,710** | ~ 60 plastic),<br>3.7 (PS) | < 10cm – 30%<br>10-50cm –67%<br>> 50cm – 3% | image analysis, Ocean Floor Observation System (OFOS), depth 2500m | Bergman and Klages, 2012 |
| 2003 | Greece, coastal waters | av. **18,400** (0-251,000) | 55.5 (plastic) | - | visual census, SCUBA divers, depth 0-25m | Katsanevakis and Katsarou, 2004 |
| 2005-05 | SE USA, Gray's Reef | av. **5,200** (400 – sparse live bottom; 9700 – ledge) | - | - | SCUBA divers,<br>depth 16-20m | Bauer *et al.*, 2008 |
| 2005 | Malta | av. **97** | 47 (plastic) | - | bottom otter trawls, 20mm mesh,<br>debris > 2cm,<br>depth ~ 50 – 700m | Mifsud *et al.*, 2013 |
| 2007 | US, Florida Keys Nat. Marine Sanctuary | **250-1,650**<br>ghost fishing traps: 20 | 12.7 (plastic) | - | manta tow diving, fishing traps collected (usually <1m), | Uhrin *et al.*, 2014 |

| | | | | | | |
|---|---|---|---|---|---|---|
| | | non-fishing traps: 254 | | | max. depth 20m | |
| 2007 | Portugal, coastal canyons | Lisbon: **6,626**, Setubal: **2,463**; Cascais: **1,058**, Nazare: **0-2,030** | 25-86 (plastic) | Majority of litter was 10-50cm | image and video analysis, ROV, depth 740-4600m | Mordecai *et al.*, 2011 |
| 2007-08 | Turkey, Black Sea coast | av. **541** (128-1,320) | 79.6 (nylon),<br>10.3 (hard pl.) | - | bottom trawl, 22mm mesh, debris >2cm, depth 25-200m | Topcu and Ozturk, 2010 |
| 2008-12 | US, Virginia, Chesapeake Bay | 4-year av. pots density **5.3**;<br>max. 15.4 (Potomac) | - | - | side imaging, derelict fishing gear (pots ~1m),<br>mostly shallow (2-6m) | Bilkovic *et al.*, 2014 |
| 2009 | Mediterranean,<br>Italy, Greece, Spain | Tyrrhenian coast: **5,999**;<br>Ionian coast: **2,518**;<br>Muricia coast: **4,305**<br>Catalan coast: **9,761** | 65-93 (plastic) | 28% - 2-5cm,<br>29% - 5-10cm,<br>32% - 10-20cm, | surface dredge, 10mm mesh,<br>debris > 2cm,<br>depth 40-80m | Sanchez *et al.*, 2013 |
| 2010-11 | Belgian coast | av. **3125** (1250-11,527) | 95.7 (plastic) | - | bottom otter and beam trawl, 10mm mesh, debris >1cm | van Cauwenberghe *et al.*, 2013 |
| 2010-11 | Atlantic, Azores, Condor seamount | Seamount summit: **1,439**<br>Northern flank: **397** | 50-73 (fishing line) | - | video analysis, ROV,<br>depth 185-1092m | Pham *et al.*, 2013 |
| 2011 | Caribbean, Bonaire, Lac Bay | **260,000-710,000** | 71 (plastic) | - | SCUBA/snorkel (?),<br>debris > 5cm | Debrot *et al.*, 2013b |
| 2012 | Turkey, Bay of Antalya | **115-2,762** | 81.1 (plastic) | - | bottom trawls, 24-44mm mesh, debris >2.4cm , depth 200-800m | Guven *et al.*, 2013 |
| Indian ocean and surrounding waters | | | | | | |
| 2003 | Jordan, Gulf of Aqaba, E coast | av. **2,800,000,** (900,000-5,900,000)<br>max. Ras Al Yamaneya 5,900,000 | 42 | - | SCUBA diving | Abu-Hilal and Al-Najjar, 2009a |
| 2006 | Jordan, Gulf of Aqaba (Al-Ghandoor) | av. **3,040,000**<br>(8,000-15,000,000) | 38 (plastic) | - | SCUBA diving | Al-Najjar and Al-Shiyab, 2011 |

### b) sea floor sediment debris (various units)

| Time | Location | Quantity | Proportion of plastic (%) | Debris size distribution | Method | Reference |
|---|---|---|---|---|---|---|
| Atlantic and surrounding waters | | | | | | |
| 2004 (?) | UK coastline, estuary and subtidal sediments | fibers per 50mL of sediment:<br>2 – estuary sediments<br>5.5 – subtidal sediments | - | - | density separation (saturated solution of NaCl) | Thompson *et al.*, 2004 |
| 2011 (?) | Belgian harbours, coast and offshore | harbour: 167 particles kg-1,<br>coastal and offshore: 97<br>beach: 93 | only microplastic included | - | Van Veen grabs, corer for beach sediments, density separation and sieving through 0.038mm mesh, debris 0.038 – 1mm | Claessens *et al.*, 2011 |
| 2013 (?) | Atlantic and Mediterranean | av. 0.5 microplastic particles per 25cm$^2$ of the sediment | only microplastic | - | top 1cm of deep sea sediments (1176-4844m), sieved 0.035-1mm | van Cauwenberghe *et* |

| | | | | | | |
|---|---|---|---|---|---|---|
| | | (max. 1 particle per 25cm$^2$ – Porcupine Abyssal Plain) | included | | mesh, density separation, debris 0.035-1mm | *al.*, 2013 |
| 2013 (?) | Adriatic, Venice Lagoon | 672 – 2175 pieces kg$^{-1}$ | only microplastic included | 93% 0.03-0.5mm | top 5cm of lagoon sediments, max. depth 1m, debris <1mm | Vianello *et al.*, 2013 |

Av. – average
Max. – maximum
Ɛ – ambiguity or possibility of error in the paper
PE - Polyethylene
PU – polyurethane
PS – polystyrene
SF – Styrofoam,
(?) – time of study not provided in the text
ROV – Remotely operated vehicle

# Appendix B

## Tables – Chapter 1.7.

Table 1.7.1. Plastic ingestion by sea birds in the Pacific region.

| Year | Location | Species | Amount ingested per bird | Percentage containing plastic | Method | Reference |
|---|---|---|---|---|---|---|
| 1980 | California, Bodega Harbour | *Phalaropus fulicarius* | av. 6.7p, max 14 (av. 0.033g, max. 0.048g) | 86% (of 7) | dead (stuck in wires) | Conors and Smith, 1982 |
| 1982, 1983 | Hawaii, Midway Atoll | *Phoebastria immutabilis* and *Puffinus pacificus* | D. immutabilis: 35.7-76.6 g,<br>P. pacificus: 2-6 fragments | 90% (albatross),<br>60% (shearwater) | unclear (dead and alive) | Fry *et al.*, 1987 |
| 1987 | NE Pacific, off British Columbia, Washington and Oregon | 8 of 11 species ingested plastic | av. < 0.3g<br>(max. 6.6g - Diomedea nigripes) | 72% (of 11 species) | bycatch (drowned) | Blight and Burger, 1997 |
| 1993-2006 | Alaska, Aleutian Islands | *Aethia sp. (aucklets)* | 1 piece in one bird | 0.0004 (1 of 2541 birds) | collecting regurgitated chick meals | Bond et al., 2010 |
| 1994-95 | Midway Atoll | *Phoebastria immutabilis* | 1994: 23.8 (dead) - 11.3 (injured)<br>1995: 18.1 (dead) - 9.5 (injured) | 97.6% (of 251) | carcasses (stranded) and euthanised injured birds | Auman *et al.*, 1997 |
| 1997-99, 2001 | Bering Sea | *Puffinus tenuirostris* | av. 6.9p (136g and 237$\text{mm}^3$) | 83.9% (of 330 short-tailed shearwaters) | lethal (shooting) | Vlietstra and Parga, 2002 (including Ogi, 1990) |
| 2001-02, 2005, 2007 | Australia, Lord Howe Island | *Puffinus carneipes*,<br>*Puffinus pacificus* | P. carneipes: 41p (20.5 $\text{cm}^3$) - dead<br>P. pacificus: av. 0.7 $\text{cm}^3$ - live | 79% (P. carneipes),<br>43% (P. pacificus) | dead, live (stomach flushing) | Hutton *et al.*, 2008 |
| 2003, 2004 | Canada, Nanavut, Cape Vera colony | *Fulmarus glacialis* | av. 7.4 | 31% (of 102) | lethal (shooting or decapitating) | Mallory, 2008 |
| 2003, 2005 | N Pacific, Bering Sea | *Puffinus tenuirostris* | av. 15.1p (0.23g) | | bycatch (drowned) | Yamashita *et al.*, 2011 |
| 2006-08 | near Hawaiian Islands | *Phoebastria immutabilis*,<br>*Phoebastria nigripes* | P. immutabilis: av. 0.99g,<br>P. nigripes: av. 0.13g | 83.3% (of 18 P. immutabilis),<br>51.7% (of 29 P. nigripes) | bycatch (drowned) | Gray *et al.*, 2012 |
| 2009-10 | E North Pacific | *Fulmarus glacialis* | av. 36.8p (0.385g),<br>max. 454p (3.66g) | 92.5% (of 67 carcasses) | salvaged at sea, stranded, collected for diet studies, bycatch | Avery-Gomm *et al.*, 2012 |
| 2010 | Australia, Victoria, Phillip Island | *Puffinus tenuirostris (fledging chicks)* | av. 7.6p (113mg) | 100% (of 67) | carcasses (stranded) | Carey, 2011 |

| | | | | | | |
|---|---|---|---|---|---|---|
| 2010, 2012 | Australia, Northstradbroke Is. | *Puffinus tenuirostris* | av. 3p (2010 - adults),<br>4.8p (2012 - juveniles) | > 67%<br>(63% of 102 adults, and 85% of juveniles) | carcasses (stranded) | Acampora *et al.*, 2014 |
| 2011 | Australlia, Lord Howe Island | *Puffinus carneipes* | av. 17.5p (2.7g) | 90% (of 38 fledgelings) | live (stomach flushing) | Lavers *et al.*, 2014 |
| 2012 | Australia, S GBR, Heron Island | *Ardenna pacifica* | 3.2p (max. 5p) | 21% (of 24 chicks only) | live (stomach flushing) | Verlis *et al.*, 2013 |

Table 1.7.2. Plastic ingestion by sea turtles in the Pacific region.

| Year | Location | Species | Percentage of sea turtles containing plastic | Amount or type of ingested debris | Method | Reference |
|---|---|---|---|---|---|---|
| 1975-2010 | Hawaii | *Chelonia mydas* | 3% (N = 2471) | monofilament line, string and plastic film | dead (stranded), live (lavage or from the mouth N = 170) | Russell *et al.*, 2011 |
| 1990-92 | Central North Pacific | *Caretta caretta* (10-80cm) | 35% (N = 52) stomach | plastic beads, sheet, PP line, small soy sauce container (little fish) | bycatch | Parker *et al.*, 2005 |
| 1990-91;<br>1999-2004 | Central North Pacific;<br>Hawaii | *Chelonia mydas* | 70% (N = 10) stomach | soft and hard plastic, PP and monofilament line | bycatch | Parker *et al.*, 2011 |
| 1995-99 | Bahia de Los Angeles | *Chelonia mydas* (50-95cm) | 1.9% (N = 101) lavage,<br>19% (N = 42) fecal,<br>29% (N = 7) stomach | plastic bags, nylon chord, .. | live animals (lavage, feces), dead (stranded and bycatch) | Seminoff *et al.*, 2002 |
| 2005-2011 | Queensland | *Chelonia mydas* (N = 88),<br>*Eretmochelis imbricata* (N = 24),<br>*Caretta caretta* (N = 2),<br>*Natator depressus* (N = 1) | All turtles: av. 39% (N = 115),<br>12% (N = 22) pelagic (< 35cm)<br>27% (N = 93) benthic (> 35cm) | av. 31.7 items per turtle;<br>34% soft plastic, 33.1% hard plastic,<br>13.6% plastic rope;<br>fishing line and packing straps | dead (stranded and deceased in a rehabilitation centre) | Schuyler *et al.*, 2012 |
| 2008 (?) | SW Pacific, N NSW and S Queensland | *Chelonia mydas, Caretta caretta* (post-hatchlings), (av. 6.4-7.7cm) | stomachs:<br>57% (N = 7), C. caretta,<br>74% (N = 34), C. mydas stranded,<br>46% (N = 13), C. mydas eaten | nylon, plastic | stranded or taken from stomachs of bycaught dolphin fish | Boyle and Limpus, 2008 |

Table 1.7.3. List of marine mammals reported to ingest marine debris globally.

| Species | IUCN status | Reference |
|---|---|---|
| Cetartiodactyla, Cetacea | | |
| Odontoceti | | |
| sperm whale<br>(*Physeter macrocephalus*) | Vulnerable | Walker and Coe, 1990, Roberts, 2003; Evans and Hindell, 2004; Fernandez *et al.*, 2009; Jacobsen *et al.*, 2010; Mazzariol *et al.*, 2011, de Stephanis *et al.*, 2013 |
| pygmy sperm whale<br>(*Kogia breviceps*) | Data deficient | Walker and Coe, 1990, Tarpley and Marwitz 1993, Stamper *et al.*, 2006, Fernandez *et al.*, 2009 |
| dwarf sperm whale<br>(*Kogia sima*) | Data deficient | Walker and Coe, 1990 |
| Baird's beaked whale<br>(*Berardius bairdii*) | Data deficient | Walker and Coe, 1990 |
| Cuvier's beaked whale<br>(*Ziphius cavirostris*) | Least Concern | Gomercic *et al.*, 2006 |
| Blainville's beaked whale<br>(*Mesoplodon densirostris*) | Data deficient | Secchi and Zarzur, 1999 |
| Gervais' beaked whale<br>(*Mesoplodon europaeus*) | Data deficient | Fernandez *et al.*, 2009 |
| common dolphin<br>(*Delphinus Delphi*) | Vulnerable | Walker and Coe, 1990 |
| Franciscana dolphin<br>(*Pontopria blainvillei*) | Vulnerable | Denuncio *et al.*, 2011, di Beneditto and Ramos, 2014 |
| harbour porpoise<br>(*Phocoena phocoena*), | Least Concern | Kastelein and Lavaleije, 1992; Baird and Hooker, 2000; Tonay *et al.*, 2007 |
| Dall's porpoise<br>(*Phocoenoides dalli*) | Least Concern | Walker and Coe, 1990 |
| rough-toothed dolphin<br>(*Steno bredanensis*) | Least Concern | de Meirelles and Barros, 2007 |
| common bottlenose dolphin (*Tursiops truncatus*) | Least Concern | Levy *et al.*, 2009 |
| Fraser's dolphin<br>(*Lagenodelphis hosei*) | Least Concern | Fernandez *et al.*, 2009 |
| striped dolphin<br>(*Stenella coerueloalba*) | Least Concern | Fernandez *et al.*, 2009 |
| Pacific white-sided dolphin<br>(*Lagenorhynchus obliquidens*) | Least Concern | Walker and Coe, 1990 |
| Risso's dolphin<br>(*Grampus griseus*) | Least Concern | Walker and Coe, 1990 |
| northern right whale dolphin<br>(*Lissodelphis borealis*) | Least Concern | Walker and Coe, 1990 |
| Guiana dolphin | Data deficient | di Beneditto and Ramos, 2014 |

| | | |
|---|---|---|
| (*Sotalia guianensis*) | | |
| short-finned pilot whale (*Globicephala macrorynhus*) | Data deficient | Fernandez *et al.*, 2009 |
| Mysticeti | | |
| fin whale (*Balenoptera physalus*) | Endangered | Fossi *et al.*, 2012, Baulch and Perry 2014 |
| sei whale (*Balenoptera borealis*) | Endangered | Baulch and Perry, 2014 |
| minke whale (*Balenoptera acutorostrata*) | Least concern | Tarpley and Marwitz 1993, Gerard 2002 |
| gray whale (*Eschrichtius robustus*) | Least concern | Cascadia research, 2010 |
| Bryde's whale (*Balaenoptera edeni*) | Data deficient | Townsville Bulletin, 2001, in Simmonds 2012, p. 5 |
| Sirenia | | |
| Florida manatee (*Trichechus manatus latirostris*) | Endangered | Beck and Barros, 1991 |
| Carnivora, Pinnipedia | | |
| Stellar sea lion (*Eumetopias jubatus*) | Near threatened | Raum-Suryan *et al.*, 2009 |
| Subantarctic fur seal (*Arctocephalus tropicalis*) | Least concern | Eriksson and Burton, 2003 |
| antarctic fur seal (*Arctocephalus gazella*) | Least concern | Eriksson and Burton, 2003 |
| harbour seal (*Phoca vitulina*) | Least concern | Rebolledo *et al.*, 2013 |

Table 1.7.4. Marine debris ingestion by marine mammals in the Pacific region.

| Year | Location | Species | Percentage of marine mammals containing plastic | Amount or type of ingested debris | Method | Reference |
|---|---|---|---|---|---|---|
| 1963-64 | New Zealand | Physeter macrocephalus | N/A | 2 slats of unwaterlogged box wood in stomach and esophagus | whaling | Gaskin and Cawthorn, 1967 |
| 1963-86 | E and W North America, Gulf of Mexico, Hawaii | 16 odontocete species | N = 43 | plastic bags and sheetings most common (62.5%) | stranded | Walker and Coe, 1990 |
| 1990/91, 1996/97 | Australia, Macquarie Island | Arctocephalus tropicalis, Arctocephalus gazella | 164 pieces of plastic in 145 seal scats (1.13p/scat) | irregular plastic fragments | fur seal feces (scats) | Eriksson and Burton, 2003 |

| | | | | | | |
|---|---|---|---|---|---|---|
| 1998 | Australia, Tasmania | Physeter macrocephalus | 11% (N = 36) | small plastic pieces | stranded | Evans and Hindell, 2004 |
| 2008 | US, Northern California | Physeter macrocephalus | N = 2 | 59 and 105 pieces of nets in two whales, plus lines and bags (weight > 24 and 73kg of debris, resp.) | stranded | Jacobsen *et al.*, 2010 |

Table 1.7.5. List of fish species reported globally to ingest plastic debris.

| Family | Species | Amount of rubbish (p/fish or mg/fish) or ingestion rate (%) | Location | Feeding and habitat | IUCN status | Reference |
|---|---|---|---|---|---|---|
| Class Actinopterygii | | | | | | |
| Alepisauridae | Longnosed lancefish<br>*Alepisaurus ferox* | * 30% (43/144)<br>° 24.5% (47/1920 | * N and central Pacific<br>° NE Pacific | Mesopelagic piscivorous predator | LC | Choy and Drazen 2013 *<br>Jantz *et al.*, 2013 ° |
| Ariidae | Madamango sea catfish<br>*Cathorops spixii* | 18% (11/60) | Goiana Estuary, NE Brazil | Estuarine | NE | Possatto *et al.*, 2011 |
| | Raspfin sea catfish<br>*Cathorops agassizii* | 33.3% (20/60) | Goiana Estuary, NE Brazil | Estuarine | NE | Possatto *et al.*, 2011 |
| | Flapnose sea catfish<br>*Sciades herzbergii* | 17.8% (11/62) | Goiana Estuary, NE Brazil | Estuarine | NE | Possatto *et al.*, 2011 |
| Atherinopsidae | Silversides<br>*Menidia menidia* | 33% (min. 5 examined) | Niantic bay, Connecticut | - | LC | Carpenter *et al.*, 1972 |
| Callionymidae | Dragonet<br>*Callionymus lyra* | 38% (19/50) | English Channel | Demersal | NE | Lusher *et al.*, 2013 |
| Carangidae | Horse mackerel<br>*Trachurus trachurus* | * 1%<br>° 28.6% (16/56) | * North Sea<br>° English Channel | Pelagic | NE | Foekema *et al.*, 2013 *<br>Lusher *et al.*, 2013 ° |
| Cepolidae | Redband fish<br>*Cepola macrophthalma* | 32.3% (20/62) | English Channel | Demersal | NE | Lusher *et al.*, 2013 |
| Coryphenidae | Common dolphinfish<br>*Coryphaena hippurus* | 2% (1/42) | N and central Pacific | Epipelagic predator | LC | Choy and Drazen, 2013 |
| Cottidae | Grubby<br>*Myxocephalus aenaeus* | 4.2% (min. 5 examined) | Niantic bay, Connecticut | - | NE | Carpenter *et al.*, 1972 |
| Gadidae | Pollack<br>*Pollachius sp.* | - | English Channel | - | NE | Anon, 1975 |
| | Cod<br>*Gadus morhua* | * 13%<br>° - | * North Sea<br>° English Channel | Demersal | VU | Foekema *et al.*, 2013 *<br>Anon 1975 ° |
| | Poor cod<br>*Trisopterus minutus* | 40% (20/50) | English Channel | Pelagic | LC | Lusher *et al.*, 2013 |
| | Pout whiting<br>*Trisopterus luscus* | - | English Channel | - | NE | Anon, 1975 |
| | Whiting | * 6% | * North Sea | Pelagic | NE | Foekema *et al.*, 2013 * |

| | | | | | | |
|---|---|---|---|---|---|---|
| | *Merlangius merlangus* | ° 32% (16/50) | ° English Channel | | | Lusher *et al.*, 2013 ° |
| | Blue whiting<br>*Micromesistius poutassou* | 51.9% (14/27) | English Channel | Pelagic | NE | Lusher *et al.*, 2013 |
| | Haddock<br>*Melanogrammus aeglefinus* | 6% | North Sea | Benthic predator | VU | Foekema *et al.*, 2013 |
| Gempylidae | Snake mackerel<br>*Gempylus serpens* | <1% (1/104) | N and central Pacific | - | NE | Choy and Drazen, 2013 |
| Gerreidae | Brazilian mojarra<br>*Eugerres brasilianus* | 16.3% (39/240) | Goiana Estuary, NE Brazil | Estuarine | NE | Ramos *et al.*, 2012 |
| | Flagfin mojarra<br>*Eucinostomus melanopterus* | 9.2% (13/141) | Goiana Estuary, NE Brazil | Estuarine | NE | Ramos *et al.*, 2012 |
| | Caitipa mojarra<br>*Diapterus rhombeus* | 11.4% (5/44) | Goiana Estuary, NE Brazil | Estuarine | NE | Ramos *et al.*, 2012 |
| Gobiidae | Sand goby<br>*Pomatoschistus minutus* | 1974-75: nil-25%<br>(0/100 – 12.5/50) | Severn Estuary and Bristol Channel | Estuarine | NE | Kartar *et al.*, 1973 |
| Lampridae | Big-eye and small-eye moonfish<br>*Lampris sp.* | Big-eye: 43% (49/115)<br>Small-eye: 58% (14/24) | N and central Pacific | Mesopelagic predator | NE | Choy and Drazen, 2013 |
| Liparidae | Sea-snail<br>*Liparis liparis liparis* | 1973-74: nil-6.5%<br>(0/100 – 3.25/50) | Severn Estuary and Bristol Channel | - | LC | Kartar *et al.*, 1976 |
| Lotidae | Five-bearded rockling<br>*Ciliata mustela* | 1974-74: 0-10%<br>(0/5-5/50) | Severn Estuary and Bristol Channel | - | NE | Kartar *et al.*, 1976 |
| Moronidae | White perch<br>*Morone americana* | 33% (min. 5 examined) | Niantic bay, Connecticut | - | LC | Carpenter *et al.*, 1972 |
| Myctophidae | Bigfin lantern fish<br>*Symbolophorus californiensis* | 7.2p/fish, 5.21mg/fish | N Pacific Subtropical Gyre | Mesopelagic | NE | Boerger *et al.*, 2010 |
| | Golden lantern fish<br>*Myctophum aurolaternatum* | 6p/fish, 4.66mg/fish | N Pacific Subtropical Gyre | Mesopelagic | NE | Boerger *et al.*, 2010 |
| | Pearly lantern fish<br>*Myctophum nitidulum* | 16% (4/25) | N Pacific Subtropical Gyre | Mesopelagic | NE | Davison and Asch, 2011 |
| | *Loweina interrupta* | 1p/fish, 0.64mg/fish | N Pacific Subtropical Gyre | Mesopelagic | NE | Boerger *et al.*, 2010 |
| | Reinhardt's lantern fish<br>*Hygophum reinhardtii* | 1.3p/fish, 1.82mg/fish | N Pacific Subtropical Gyre | Mesopelagic | NE | Boerger *et al.*, 2010 |
| | Andersen's lantern fish<br>*Diaphus anderseni* | 15.4% (2/13) | California current and N Pacific Subtropical Gyre | Mesopelagic | NE | Davison and Asch, 2011 |
| | *Diaphus fulgens* | 28.5% (2/7) | California current and N Pacific Subtropical Gyre | Mesopelagic | NE | Davison and Asch, 2011 |
| | Bolin's lantern fish<br>*Diaphus phillipsi* | 100% (1/1) | California current and N Pacific Subtropical Gyre | Mesopelagic | NE | Davison and Asch, 2011 |
| | Cocco's latern fish<br>*Lobianchia gemellarii* | 33.3% (1/3) | California current and N Pacific Subtropical Gyre | Mesopelagic | NE | Davison and Asch, 2011 |
| Pleuronectidae | European flounder<br>*Platichthys flesus* | 1972-1974: nil-20.7%<br>(0/200 – 110/530), | Severn Estuary and Bristol Channel | Estuarine | LC | Kartar *et al.*, 1976 |
| | Winter flounder | 2.1 % (min. 5 examined) | Niantic bay, Connecticut | - | NE | Carpenter *et al.*, 1972 |

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|---|---|---|---|---|---|---|
| | *Pseudopleuronectes americanus* | | | | | |
| Sciaenidae | Drum<br>*Stellifer brasiliensis* | 6.9% (23/330) | Goiana Estuary, NE Brazil | Demersal, estuaries | NE | Dantas *et al.*, 2012 |
| | Drum (Little croaker)<br>*Stellifer stellifer* | 9.2% (22/239) | Goiana Estuary, NE Brazil | Demersal, estuaries | NE | Dantas *et al.*, 2012 |
| Scombridae | Bigeye tuna<br>*Thunnus obesus* | 9% (3/35) | N and central Pacific | - | VU | Choy and Drazen, 2013 |
| Scomberesocidae | Pacific saury<br>*Cololabis saira* | 3.2p/fish, 1.97mg/fish | N Pacific Subtropical Gyre | Epipelagic | NE | Boerger *et al.*, 2010 |
| Soleidae | Solenette<br>*Buglossidium luteum* | 26% (13/50) | English Channel | Demersal | LC | Lusher *et al.*, 2013 |
| | Thickback sole<br>*Microchirus variegatus* | 23.5% (12/50) | English Channel | Demersal | NE | Lusher *et al.*, 2013 |
| Sparidae | Red sea bream<br>*Pagellus bogaraveo* | 1.7% (1/60) | E Ionian Sea, Greece | - | NE | Anastasopoulou *et al.*, 2013 |
| Sternoptychidae | Highlight hatchetfish<br>*Sternoptyx pseudobscura* | 16.7% (1/6) | California current and N Pacific Subtropical Gyre | Mesopelagic | NE | Davison and Asch, 2011 |
| Stomiidae | *Astronesthes indopacificus* | 1p/fish, 0.03mg/fish | N Pacific Subtropical Gyre | Mesopelagic | LC | Boerger *et al.*, 2010 |
| | Pacific blackdragon<br>*Idiacanthus antrostomus* | 25% (1/4) | California current and N Pacific Subtropical Gyre | Mesopelagic | NE | Davison and Asch, 2011 |
| Triglidae | Red gurnard<br>*Chelidonichthys cuculus* | 51.5% (34/66) | English Channel | Demersal | NE | Lusher *et al.*, 2013 |
| Zeidae | John Dory<br>*Zeus faber* | 47.6% (20/42) | English Channel | Pelagic | NE | Lusher *et al.*, 2013 |
| Xiphiidae | Broadbill swardfish<br>*Xiphias gladius* | 3% (1/31) | N and central Pacific | - | LC | Choy and Drazen, 2013 |
| Class Elasmobranchii | | | | | | |
| Carcharninidae | Tiger shark<br>*Galeocerdo cuvier* | 7.52% (38/505) | KwaZulu-Natal, South Africa | - | NT | Cliff *et al.*, 2002 |
| | Bull shark<br>*Carcharhinus leucas* | 0.61% (4/661) | KwaZulu-Natal, South Africa | - | NT | Cliff *et al.*, 2002 |
| | Copper shark (Bronze Whaler)<br>*Carcharhinus brachyurus* | 0.06% (1/1404) | KwaZulu-Natal, South Africa | - | NT | Cliff *et al.*, 2002 |
| | Blacktip shark<br>*Carcharhinus limbatus* | 0.22% (4/1785) | KwaZulu-Natal, South Africa | - | NT | Cliff *et al.*, 2002 |
| | Dusky shark<br>*Carcharhinus obscurus* | 0.15% (4/2741) | KwaZulu-Natal, South Africa | - | VU | Cliff *et al.*, 2002 |
| Dasyatidae | Pelagic stingray<br>*Pteroplatytrygon violacea* | 50% (1/2) | E Ionian Sea, Greece | - | LC | Anastasopoulou *et al.*, 2013 |
| Etmopteridae | Velvet-belly lanternshark<br>*Etmopterus spinax* | 6.3% (1/16) | E Ionian Sea, Greece | - | LC | Anastasopoulou *et al.*, 2013 |
| Lamnidae | Great white shark<br>*Carcharodon carcharias* | 0.38% (2/524) | KwaZulu-Natal, South Africa | - | VU | Cliff *et al.*, 2002 |

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|---|---|---|---|---|---|---|
|  | Shortfin mako shark<br>*Isurus oxyrinchus* | 0.87% (2/231) | KwaZulu-Natal, South Africa | - | VU | Cliff *et al.*, 2002 |
| Odontaspidae | Spotted ragged–tooth<br>*Carcharias taurus* | 0.04% (1/2268) | KwaZulu-Natal, South Africa | - | VU | Cliff *et al.*, 2002 |
| Rhincodontidae | Whale shark<br>*Rhincodon typus* | 100% (1/1) | Samutprakarn, Thailand | - | VU | Haetrakul *et al.*, 2007 |
| Scyliorhinidae | Blackmouth catshark<br>*Galeus melastomus* | 3.2% (24/741) | E Ionian Sea, Greece | - | LC | Anastasopoulou *et al.*, 2013 |
| Sphyrnidae | Scalloped hammerhead shark<br>*Sphyrna lewini* | 0.1% (2/1916) | KwaZulu-Natal, South Africa | - | EN | Cliff *et al.*, 2002 |
|  | Smooth hammerhead shark<br>*Sphyrna zygaena* | 0.17% (2/1154) | KwaZulu-Natal, South Africa | - | VU | Cliff *et al.*, 2002 |
| Squalidae | Longnose spurdog<br>*Squalus blainville* | 1.3% (1/75) | E Ionian Sea, Greece | - | DD | Anastasopoulou *et al.*, 2013 |

Table 1.7.6. Marine debris ingestion by invertebrates:

a) In the wild

| Taxonomy | Species | Amount of debris (p/ind or mg/ind) or ingestion rate (%) | Location | Feeding and habitat | Reference |
|---|---|---|---|---|---|
| Annelida |  |  |  |  |  |
| Polychaeta, Sedentaria | *Clymenella torquata* | - | Halifax Harbour, Nova Scotia | detrivore | Mathalon and Hill, 2014 |
| Polychaeta, Errantia | *Alitta virens* | - | Halifax Harbour, Nova Scotia | detrivore | Mathalon and Hill, 2014 |
| Arthropoda, Crustacea |  |  |  |  |  |
| Decapoda | *Nephrops norvegicus* | 83% (100/120) | N Clyde Sea, Scotland, UK | omnivore, predator, scavenger | Murray and Cowie, 2011 |
| Cirripedia | *Lepas* spp. (*Lepas (Anatifa) anatifera and Lepas pasifica*) | 33.5% (129/385) | North Pacific Sybtropical Gyre | omnivorous opportunistic feeder | Goldstein and Goodwin, 2013 |
| Mollusca |  |  |  |  |  |
| Bivalvia | *Mytilus edulis* | Wild: 34p/mussel<br>Aquaculture: 75p/mussel | Halifax Harbour, Nova Scotia | filterfeeder | Mathalon and Hill, 2014 |
| Bivalvia | *Mytilus galloprovincialis* | - | Alfacs Bay, Ebro Delta, Mussel aquaculture, Spain | filterfeeder | Galimany *et al.*, 2009 |
| Caethognata |  |  |  |  |  |
|  | *Parasagitta elegans* | 1 specimen with 1 spherule | Coastal waters of S New England, US | predator | Carpenter *et al.*, 1972 |

b) Feeding experiments

| Taxonomy | Organism | Life Stage | Reference |
|---|---|---|---|
| Annelida | | | |
| Polychaeta, Sedentaria | *Arenicola marina* | Adult | Thompson *et al.*, 2004, Besseling *et al.*, 2012, Browne *et al.*, 2013, Wright *et al.*, 2013 |
| Arthropoda, Crustacea | | | |
| Decapoda | *Nephrops norvegicus* | Adult | Murray and Cowie, 2011 |
| | Brachyura | Megalopa | Cole *et al.*, 2013 |
| | Caridea | Larva | Cole *et al.*, 2013 |
| | Paguridae | Larva | Cole *et al.*, 2013 |
| | Porcellanidae | Zoea | Cole *et al.*, 2013 |
| Amphipoda | *Orchestia gammarellus* | Adult | Thompson *et al.*, 2004 |
| | *Talitrus saltator* | Adult | Ugolini *et al.*, 2013 |
| Cirripedia | *Semibalanus balanoides* | Adult | Thompson *et al.*, 2004 |
| Copepoda | *Acartia (Acartiura) clausi* | Adult | Cole *et al.*, 2013 |
| | *Calanus helgolandicus* | Adult | Cole *et al.*, 2013 |
| | *Centropages typicus* | Adult | Cole *et al.*, 2013 |
| | *Temora longicornis* | Adult | Cole *et al.*, 2013 |
| Crustacea | Euphausiidae | Adult | Cole *et al.*, 2013 |
| Chordata | | | |
| Tunicata | Doliolidae | Adult | Cole *et al.*, 2013 |
| Tunicata, Salpida | *Thalia longicauda* | Adult | Chan and Witting 2012 |
| | *Cyclosalpa affinis* | Adult | Chan and Witting 2012 |
| | *Ritteriella amboinensis* | Adult | Chan and Witting 2012 |
| Mollusca | | | |
| Bivalvia | *Mytilus galloprovincialis* | Adult | Galimany *et al.*, 2009 |
| | *Mytilus edulis* | Adult | Browne *et al.*, 2008,<br>Von Moos *et al.*, 2012<br>Wegner *et al.*, 2012 |
| Cnidaria | | | |
| Hydrozoa | *Obelia* sp. | Adult | Cole *et al.*, 2013 |
| Echinodermata | | | |
| Holothuroidea | *Thyonella gemmata* | Adult | Graham and Thompson 2009 |
| Holothuroidea | *Holothuria (Halodeima) floridana* | Adult | Graham and Thompson 2009 |
| Holothuroidea | *Holothuria (Halodeima) grisea* | Adult | Graham and Thompson 2009 |
| Holothuroidea | *Cucumaria frondosa* | Adult | Graham and Thompson 2009 |
| Echnoidea | *Tripneustes gratilla* | Larva | Kaposi *et al.*, 2013 |
| Echnoidea | *Dendraster excentricus* | Larva | Appelmans 1994, Miner 2007 |
| Chromista | | | |
| Dinoflagellata | *Oxyrrhis marina* | - | Cole *et al.*, 2013 |

Note: Taxonomy and conservation status used from www.iucnredlist.org, www.marinespecies.org, www.fishbase.org, marinebio.org, www.catalogueoflife.org

NE – not evaluated by IUCN
DD – Data deficient
LC – Least concern
NT – Near threatened
VU - Vulnerable
EN – Endangered
CR – Critically endangered
EW – Extinct in the wild
EX – Extinct

Charles Moore in Marina Debris creation ‘Forever Moore’