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Spatio-temporal variation of anthropogenic marine debris on Chilean beaches

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ABSTRACT

We examined the hypothesis that in an emerging economy such as Chile the abundances of Anthropogenic Marine Debris (AMD) on beaches are increasing over time. The citizen science program *Científicos de la Basura* ("Litter Scientists") conducted three national surveys (2008, 2012 and 2016) to determine AMD composition, abundance, spatial patterns and temporal trends. AMD was found on all beaches along the entire Chilean coast. Highest percentages of AMD in all surveys were plastics and cigarette butts, which can be attributed to local sources (i.e. beach users). The Antofagasta region in northern Chile had the highest abundance of AMD compared with all other zones. Higher abundances of AMD were found at the upper stations from almost all zones. No significant tendency of increasing or decreasing AMD densities was observed during the 8 years covered by our study, which suggests that economic development alone cannot explain temporal trends in AMD densities.

1. Introduction

Anthropogenic marine debris (AMD), mostly composed of plastics, has been characterized as a global and persistent environmental problem that has considerably increased over the last decades (Galgani et al., 2015). The accumulation of AMD has multiple impacts that can be categorized into three types: biological, ecosystemical, and socio-economical (Kuo and Huang, 2014). Biologically, marine organisms can be affected by AMD through ingestion and entanglement (Gregory, 2009; Kühn et al., 2015), which can lead to physical and chemical damage and even death (Rochman et al., 2013; Vegter et al., 2014). In terms of ecosystems and biogeography, AMD may serve as dispersal vehicle for invasive species, which can modify local ecosystems (Amaral-Zettler et al., 2015; Kiessling et al., 2015; Thiel and Gutow, 2005). With respect to socio-economic factors, since coastal AMD is aesthetically detrimental, it can affect the perception of beach-users and eventually coastal tourism, requiring high beach cleaning costs and causing losses in revenue (Jang et al., 2014; Newman et al., 2015; Santos et al., 2005).

Marine debris can be classified according to two main source categories: ocean- and land-based debris, depending on where AMD entered the marine environment (Galgani et al., 2015). Ocean-based AMD sources include waste from shipping, fishing, oil platforms, and aquaculture (Astudillo et al., 2009; Edyvane et al., 2004; Hinojosa and Thiel, 2009; Hong et al., 2014; Sheavly and Register, 2007; Watters et al., 2010). Land-based AMD might enter the marine environment by rivers, outflow from industries, harbors, unmanaged landfills, sewage waters, extreme events (tsunamis, hurricanes), or through direct littering by beach visitors (Aguilera et al., 2016; Goto and Shibata, 2015; Green et al., 2015; Khordagui and Abu-Hilal, 1994; Rech et al., 2014). It has been estimated that > 8.4 million tons of plastic waste enter the oceans annually from land-based sources (Jambeck et al., 2015).

A major part of AMD studies have been conducted in coastal areas, because of their proximity to sources, facility of access, and due to potential aesthetic impacts (Galgani et al., 2015). These studies provide insights about dominant types of AMD, their abundance, potential origins and accumulation rates. Nevertheless, these characteristics might be highly variable, e.g. due to the influence of urban cities, coastal use, river flushing, and the geomorphology, hydrodynamics and oceanography of each coastal area (Galgani et al., 2015; Moriarty et al., 2016). Consequently, studies with standardized methods that extend over a certain region and time are needed to evaluate the spatio-

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temporal variability of AMD abundances and composition (Browne et al., 2015; Hong et al., 2014; Moreira et al., 2015; Thornton and Jackson, 1998).

Temporal changes of AMD have been studied over different spatial scales, but most of them have been done on a small scale, such as single beaches, bays or islands (e.g. Agustin et al., 2015; Tourinho and Fillmann, 2011), or during shorter periods of time (e.g. Madzena and Lasiak, 1997; Storrier et al., 2007). Only few studies extend over a large temporal and spatial scale (for examples see Gago et al., 2014; Nelms et al., 2016; Ribic et al., 2010, 2011; Schulz et al., 2015a, 2015b). The generated results are contrasting: in some cases trends of AMD densities can be identified, either increasing in Indonesia (Willoughby et al., 1997), decreasing in Australia (Edyvane et al., 2004), or being consistent over time in Canada and Scotland (Lucas, 1992; Storrier et al., 2007). But in other cases, the results are highly variable and no temporal tendencies are observed (Cheung et al., 2016; Eriksson et al., 2013; Nelms et al., 2016). Consequently, no consistent global long-term trend has been identified for AMD densities on beaches (Browne et al., 2015).

Changes of AMD abundance over time might be related to the socioeconomic and educational conditions of a country; pro-environmental behavior (e.g. adequate waste disposal) is generally higher in developed countries (Morren and Grinstein, 2016). Developed economies might have educational programs to increase environmental awareness in the population, such as environmental campaigns, frequent clean-up campaigns (e.g. NOAA, Ocean Conservancy), and well-established recycling infrastructure (Borja et al., 2011; Veiga et al., 2016). On the other hand, developing countries might focus on economic growth, often at the expense of environmental care. Environmental degradation might increase as the economy grows, up to a point where economic development leads to higher environmental awareness, and amounts of AMD are likely to be lower on beaches of developed countries, with a tendency to decrease over time. For an emerging economy such as Chile we should thus expect that the amounts of AMD of beaches are high, with a tendency to increase over time.

The Chilean citizen science program *Científicos de la Basura* ("litter scientists") has been studying the problem of AMD in the SE Pacific for approximately 10 years, conducting the first national survey of AMD on beaches in 2008 (for further details of the program, see Eastman et al., 2014). The mean density of AMD was by that time 1.8 items m^{-2} (which included wood and "others") and the most common AMD types were plastics, cigarette butts and glass (Bravo et al., 2009). Nevertheless, these results represented only a first snapshot of the general situation, and there was no information on the spatio-temporal dynamics of beach litter in the SE Pacific. Therefore, the national AMD survey was repeated during the years 2008, 2012 and 2016 along the entire Chilean coast, in order to (1) determine composition, (2) estimate abundance and spatial patterns, and (3) explore temporal trends of AMD densities on beaches from the SE Pacific.

2. Materials and methods

2.1. Study area

The Chilean coast covers most of the South East Pacific, extending over ~4500 km from 18°S up to the southern tip of the continent at 56°S. The coast is composed mainly of rocky shores and sandy beaches between 18° and 42°S, and extensive channels and archipelagos in the Patagonian region south of ~42°S (Miloslavich et al., 2011). In order to guarantee a representative sampling of the Chilean coast, we invited participants from specific locations that are more or less evenly distributed along the coast. During each AMD survey between 29 and 40 beaches were sampled, with a total of 69 different beaches for the three surveys (Fig. 1a). Locations from the oceanic islands, Rapa Nui and Juan Fernandez, were only included during the AMD survey of 2016. At each selected location, beaches were chosen depending on the importance of the beach for the community, ease of access, and proximity to the participating schools. The selected beaches ranged from urban to semi-urban beaches with sandy sediments.

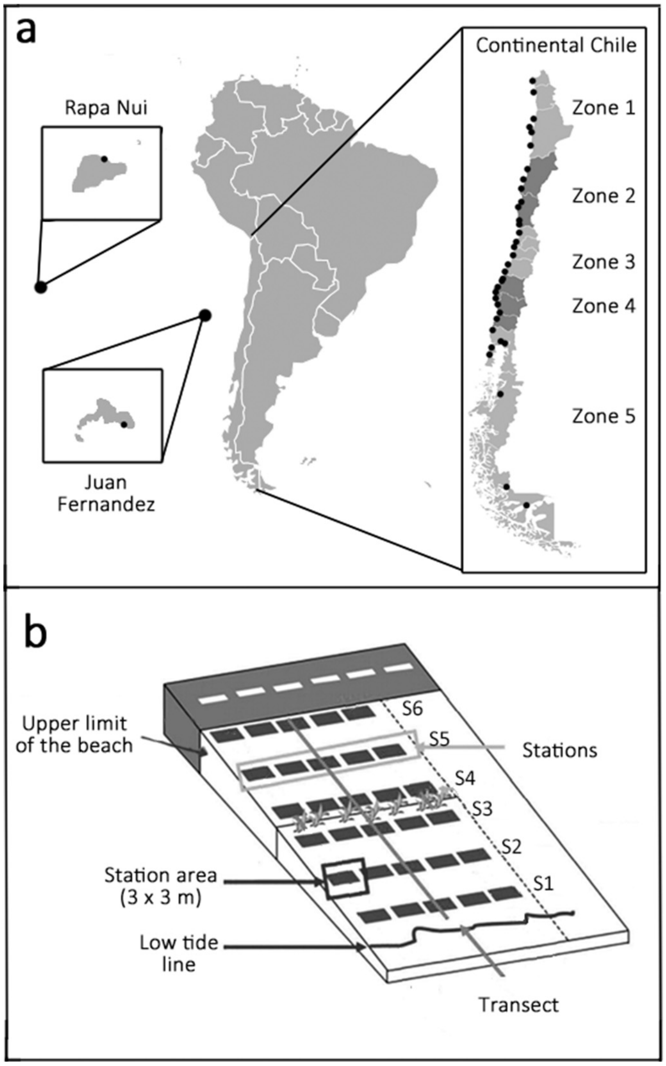

Fig. 1. (a) Beaches sampled during the three AMD surveys along the Chilean continental coast (within their five zones) and islands. (b) Schematic overview of beach survey design. S1 to S6 indicate the number of station per transect, in which station 1 (S1) represents the station closer to the low tide line, and station 6 (S6) corresponds to the station closer to the highest site of the beach.

2.2. Citizen science participation

2.2.1. Recruitment

The general coordinator of the program contacted and invited prospective schools from selected locations with potential interest in the project. Once each institution agreed to participate, one specific person acted as the responsible local person (teacher or school director) who was in charge of keeping contact with the general coordinator of the Program. Whenever possible, the schools were supported by local city governments, corporations, and local environmental organizations, in terms of transportation and materials for the activity. A total of 3551 students and 84 teachers from 99 schools participated in the three surveys, with the support of 46 regional advisors (some of which participated in two or all three survey years) (Table 1). Surveys were conducted during August–September of 2008, April–May of 2012, and June–July of 2016, i.e. during austral fall or winter.

Table 1
Number of participants per national survey.

Participants	2008	2012	2016	Total
Schools and organizations	48	26	40	99[a]
Students	1590	815	750	3155
Teachers	23	28	47	84[a]
Advisors	16	17	19	46[a]

[a] Please note that the sum between columns does not match the total amount, because there are schools, teachers and advisors who participated in more than one survey.

2.2.2. *Training activities*

A package of support materials was sent to each school. It included activity booklets, motivational story books for students, and materials for the survey (see Eastman et al., 2014). Several preparatory activities were conducted for children to learn about the ocean and marine life, and also about the problem of litter. Prior to each sampling, a dry run of the AMD survey was done with all participants, in which all steps of the sampling protocol were revisited. Schools that requested or needed support in the beach survey were matched with local marine biologists and professionals related to natural sciences; if possible, these local scientific advisors also participated in the preparatory activity. During the whole process, the general coordinator of the program was in contact with the responsible person from each school in order to ensure the correct understanding of the protocol and an effective sampling procedure.

2.3. *AMD surveys*

Between one and six transects, perpendicular to the coastline, were surveyed at each beach. Transects ran from the low tide line (Station 1) to the highest site of the beaches (basis of dunes or the beginning of a road, Station 6). Each transect had between two and six stations, depending on the width of the beach. Each station covered a quadrat of 3 m × 3 m (9 m^2) (Fig. 1b), delimited by ropes and sticks. Thus, sampling effort is standardized per area and there is no risk that the number of participants could affect the amounts of sampled AMD; instead, more participants allow for a larger number of standardized samples thereby enhancing the representativeness of the study.

All AMD items found at each station were collected and counted after classifying them according to the following categories: papers, cigarette butts, plastics, metals and glass. Original datasheets considered also wood and “others”, but these categories were not coherent among the three surveys, since in some cases natural debris (natural wood, feathers, algae) was accidentally counted by the participating schools as AMD, and consequently we decided to exclude these categories for analysis.

The information was gathered on data sheets, which were sent by the responsible teacher via website (http://www.cientificosdelabasura.cl/), e-mail, or through regular mail to the general coordination of the program at Universidad Católica del Norte in Coquimbo.

2.4. *Data evaluation and statistical analyses*

For the purpose of data evaluation, the country was divided in five large zones, according to climatic, geographic, and socio-economic factors (see Bravo et al., 2009). Rapa Nui Island and Juan Fernandez Archipelago were considered as a separate zone (oceanic islands). We compared the abundance of AMD among the three surveys (items per m^2), the composition of AMD according to the main types (% per type), and differences in abundance between the high and low stations of the beach (items per m^2).

Differences between zones and years in the proportion of AMD types were tested using two-way contingency tables. To evaluate differences among zones, three independent tables (one per year) were conducted with the zones 1–5 for the years 2008, 2012, and 2016 (including Juan Fernandez and Rapa Nui in 2016) and AMD types as variables. In a separate analysis, years (2008, 2012 and 2016) and the total amount of different types of AMD were considered as variables. The interactions between years and zones were not considered in these analyses to simplify the results and interpretation. However, a graphical representation of all data is reported.

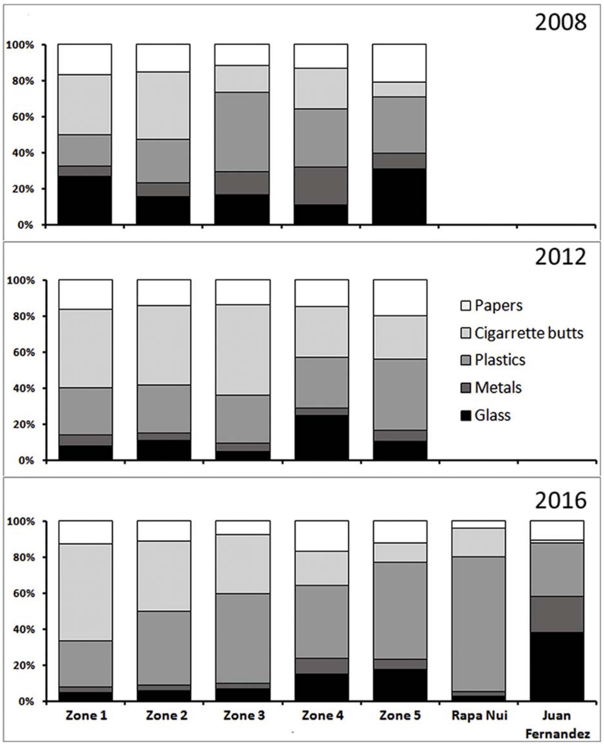

Fig. 2. Percentages of the different categories of AMD for the three surveys, in each of the five zones of Continental Chile and the oceanic islands.

To determine whether AMD abundances varied among zones over years, a generalized linear model (GLM) was applied. The GLM permits that the response variables (e.g. AMD abundance) have error distributions other than a normal distribution and unequal variances (Zuur et al., 2009). Different models with different error distributions and data transformations were run in preliminary tests to select the model providing the best fit of the data. Model selection was based on the ratio between the residual deviance and the residual degrees of freedom (Zuur et al., 2009). The best model selected was with $\log(x + 1)$ transformed data and the “quasi-poisson” distribution of the error. Zones and years were considered as fixed factors with interactions in the final model.

To determine whether AMD abundances varied between the high and low stations of the beaches a linear mixed-effects model (LME) was used, considering a nested design where the transects per zones were considered as a random factor (Zuur et al., 2009). The GLM and LME were tested using R (R Development Core Team, 2008).

3. Results

3.1. *Types of AMD*

Among all the surveys, the highest percentages of AMD were plastics (27.1% in 2008) and cigarette butts (38.0% in 2012, 41.8% in 2016). In 2008, there were significant differences among zones ($\chi^2 = 94.761$; $p < 0.001$; Fig. 2; Table S1), where cigarette butts dominated in zones 1 (33.3%) and 2 (37.5%), and plastics in zones 3 and 5 ($> 31.1\%$). In 2012, there were also significant differences between zones ($\chi^2 = 32.804$; $p = 0.008$; Fig. 2; Table S1). Cigarette butts

Table 2
Generalized linear model (GLM; quasi-poisson: Data = Log (x + 1)) results table for the abundance of AMD across zones and survey years. Significant values in bold.

Factors	Source of variation	D.F.	Deviance	d.f.	Residual	p
Years	NULL			2733	795.36	
Zones	Years	2	4.708	2731	792.66	< **0.001**
	Zones	4	67.600	2727	723.06	< **0.001**
	Years:zones	8	40.133	2719	682.92	< **0.001**

Linear hypotheses		Estimate	Std. error	z value	Pr(> |z|)
Years	2008–2012	− 0.236	0.108	− 2.188	0.073
	2008–2016	− 0.131	0.099	− 1.310	0.389
	2012–2016	0.105	0.105	1.008	0.572
Zones	Zone 1–zone 2	− 1.159	0.100	− 11.548	< **0.001**
	Zone 1–zone 3	− 1.223	0.098	− 12.521	< **0.001**
	Zone 1–zone 4	− 1.076	0.109	− 9.870	< **0.001**
	Zone 1–zone 5	− 0.440	0.089	− 4.937	< **0.001**
	Zone 2–zone 3	− 0.064	0.094	− 0.680	0.960
	Zone 2–zone 4	0.083	0.106	0.783	0.935
	Zone 2–zone 5	0.719	0.086	8.359	< **0.001**
	Zone 3–zone 4	0.148	0.104	1.422	0.610
	Zone 3–zone 5	0.783	0.083	9.451	< **0.001**
	Zone 4–zone 5	0.635	0.096	6.625	< **0.001**

Simultaneous tests for general linear hypotheses over the factor interaction represented in Fig. 3 and the Supplementary Table S1. A multiple comparisons of means using the Tukey contrasts are reported. Pr represents an adjusted p values on a single-step method.

reached the highest proportions in zones 1 to 3 (> 43.6%), and plastics in zone 5 (39.8%). Glass presented a higher proportion in zone 4 than in the other zones (24.8%). In 2016 the significant differences between zones ($\chi^2 = 121.993$; $p < 0.001$; Fig. 2; Table S1) were due to the dominance of cigarette butts in zone 1 (53.8%), and plastics in zones 2 to 5 (> 40.3%). In the case of the islands, Rapa Nui presented a high predominance of plastics (75.3%), in contrast to Juan Fernandez, where glass (38%) and plastics (30%) were predominant types of AMD (Fig. 2; Table S1).

3.2. AMD abundance

AMD was found along the entire Chilean coast. The average amount per survey of AMD was 1.37 (2008), 1.69 (2012), and 2.15 (2016) AMD items per m^2. There were statistical differences between years ($p < 0.001$) and zones ($p < 0.001$) and the interaction between these factors was also significant ($p < 0.001$) (Table 2). However, differences among years were not reflected in the multiple comparisons test and this result needs to be considered with caution (Table 2).

Table 3
Linear mixed-effects model (LME; Data = Log (x + 1)) results table for the abundance of AMD across zones and survey years; transects were considered as a random factor (nested design).

Factors	D.F.	Sum Sq	DenDF	F-value	p (> F)
Years	2	0.328	554.8	2.839	0.059
Zones	4	4.258	532.3	18.417	< **0.001**
High/low	1	4.395	558.3	76.024	< **0.001**

Linear hypotheses	Estimate	Std. error	z value	Pr(> |z|)
High - low	− 0.130	0.015	− 8.719	< **0.001**

There were some differences between years within zones (Table S2). In Zone 1 there were no differences among years. In Zone 2, Zone 3 and Zone 5, AMD densities in 2008 were lower than in 2012 and 2016. In Zone 4 higher AMD abundances were found in 2012 compared with 2016. The multiple comparisons among zones showed that zone 1 had the highest abundance of AMD compared with all other zones ($p < 0.001$), and zone 5 had higher abundances than zones 2 ($p < 0.001$), 3 ($p < 0.001$) and 4 ($p < 0.001$) (Table 2). In 2008, zone 1 had the highest abundance of AMD, and zone 5 had higher abundances of AMD compared with zones 2, 3 and 4 (Table S2). In 2012, zone 1 had higher abundances than zones 2, 4 and 5, and in 2016 the difference was found between zones 1 and all the other zones (Table S2) (Fig. 3).

Overall, abundances of AMD were higher in Antofagasta (zone 1) in comparison to the rest of the country, with 8.7 (2008), 11.4 (2012) and 13.0 (2016) AMD items per m^2. All other regions presented average values < 3.8 AMD items per m^2 during the three surveys. In the case of the oceanic islands Rapa Nui and Juan Fernandez, litter abundances were 0.5 and 0.7 AMD items per m^2, respectively (Fig. 3).

3.3. AMD at different beach stations

With respect to the differences between high and low stations, higher abundances of AMD were found in the upper areas of the beaches throughout the country ($p < 0.001$) (Table 3). Exceptions were

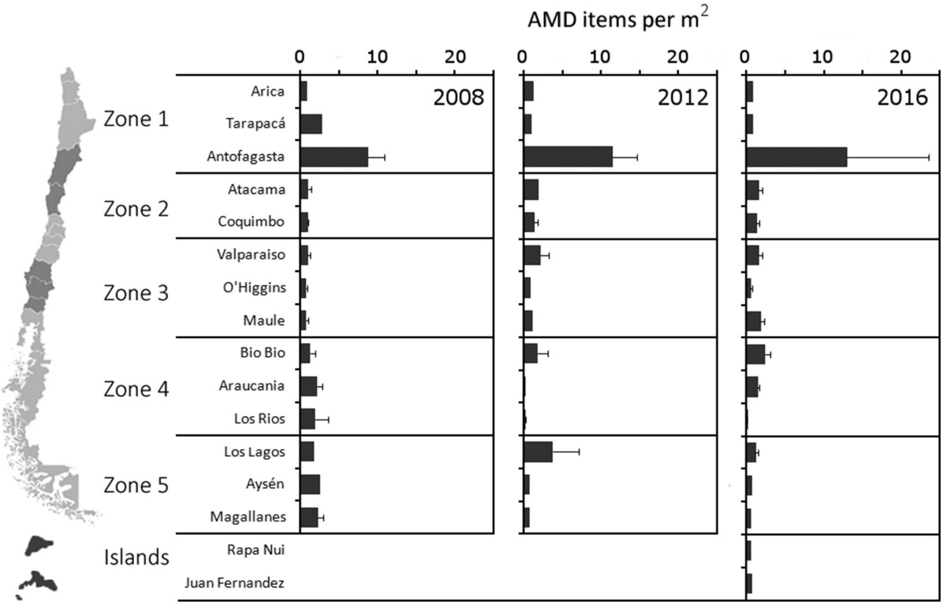

Fig. 3. AMD Abundance along the Chilean coast during the three surveys.

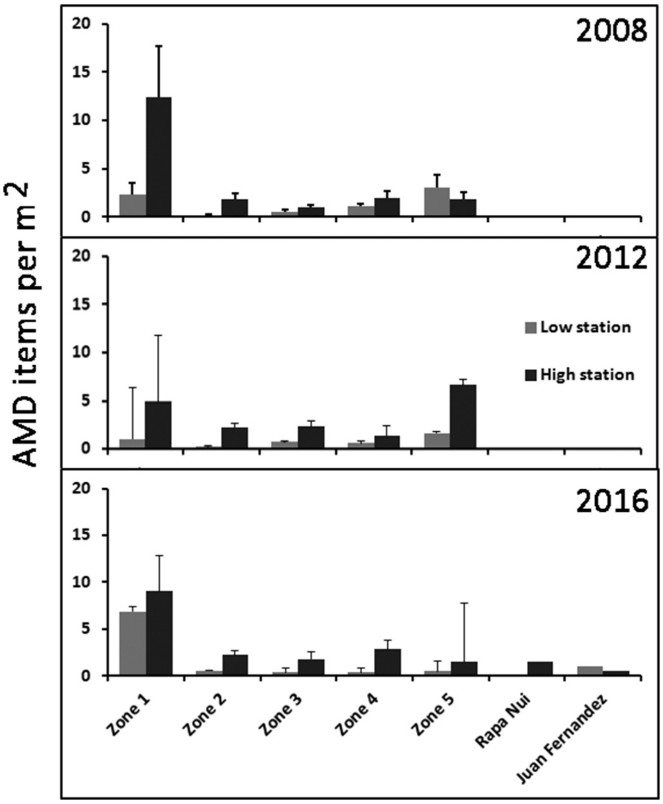

Fig. 4. AMD Abundance at the low and high stations in the five zones along the Chilean coast, Rapa Nui and Juan Fernandez Archipelago.

observed in zone 5 during 2008 and on Juan Fernandez in 2016, where higher amounts of AMD were found at the lower stations (Fig. 4).

4. Discussion

4.1. Composition of AMD

Plastics and cigarette butts constitute the larger part of AMD on Chilean beaches during all three surveys. These items can be attributed to local sources, since they are common products used by beach users (Andrades et al., 2016; Ivar do Sul and Costa, 2007; Liu et al., 2013; Rosevelt et al., 2013). Because of their durability, these items made from synthetic materials comprise a large proportion of AMD (Barnes et al., 2009; Moore, 2008). The persistent abundance of cigarette butts and small-sized plastics are hard to manage since they might be buried in the sand below the surface, and are a difficult target for beach cleanups because of their size (Portz et al., 2011). Smaller plastic items are generated by the degradation of larger items, and so they represent the accumulation of fragmented items over time (Nelms et al., 2016).

The high proportion of glass and metal pieces on Chilean beaches can also be an indicator of AMD sources. Since fragmented items from both materials do not float in seawater and are persistent materials, their presence usually indicates nearby sources, typically beach users (land-based AMD) (Bravo et al., 2009). On the other hand, plastics can indicate distant sources, such as ships and the maritime industry (ocean-based AMD) (Morishige et al., 2007). This is particularly relevant for islands such as Juan Fernandez Island, which presented a higher abundance of plastics (and glass), similar as observed on St. Lucia and Dominica (Corbin and Singh, 1993), Pitcairn Islands (Benton, 1995), and French Frigate Shoals, Hawaii (Morishige et al., 2007). Rapa Nui Island was mainly dominated by plastics. Since it is located close to the center of the South Pacific Gyre, a large fraction of these plastics come from ocean-based AMD (mostly industrial fisheries – Kiessling et al., 2017), or through their transport from continental coasts via oceanic currents (Eriksen et al., 2013; Martinez et al., 2009; Miranda-Urbina et al., 2015).

4.2. Spatial distribution and abundance of AMD

The accumulation of AMD is a prevalent problem across the Chilean continental coast and oceanic islands. In general terms, the distribution of AMD in coastal areas is highly variable (Galgani et al., 2015; Turra et al., 2014). These differences can be caused by several factors. In the first place, the proximity to sources will determine the amounts of AMD found. For instance, urbanized beaches tend to have highest AMD densities in comparison to beaches with lower population densities (Andrades et al., 2016; Nelms et al., 2016). Nevertheless, this relationship is not always straightforward, because in some cases the amount of AMD found in highly populated areas can be lower than in remote areas (e.g. Ribic et al., 2010). Most of the material found in urbanized areas comes from local sources, either from recreational use (Storrier and McGlashan, 2006; Walker, 1997), sewage dumping (Velander and Mocogni, 1998), landfills (Browne et al., 2015), or fishing activity (Claereboudt, 2004). In our study, we have identified a strong and persistent trend of AMD accumulation on beaches of the Antofagasta Region. Moreover, it has been reported that within this region, beaches close to the urban center of Antofagasta accumulate more litter than beaches from remote locations of the Antofagasta region (Kiessling et al., 2017), which highlights the local sources of AMD in this region. That same study reported that AMD on beaches is not perceived as a main issue for local government and residents, which hampers efficient AMD prevention and management (Kiessling et al., 2017).

On the other hand, the continental AMD distribution can be related to environmental and climatic factors, such as currents, waves and wind (Browne et al., 2010; Bowman et al., 1998; Rosevelt et al., 2013; Santos et al., 2009; Storrier et al., 2007; Thornton and Jackson, 1998; Williams and Tudor, 2001). For instance, protected bays might accumulate higher abundances of AMD, compared to beaches with direct exposition to wind (Ivar do Sul et al., 2009; Li et al., 2016). Wind also affects AMD densities, since light AMD materials (e.g. expanded polystyrene) accumulate downwind, in contrast to denser materials that are dispersed more evenly across the beach or estuary extension (Browne et al., 2010). Proximity to rivers can also increase AMD on nearby beaches, since they can transport AMD via storm-water runoff or direct deposition (Araújo and Costa, 2007; Rech et al., 2014; Sadri and Thompson, 2014), which might be intensified during El Niño years (Thiel et al., 2013).

Oceanic currents and proximity to the subtropical gyres are important factors of AMD accumulation on oceanic islands (Blickley et al., 2016; Goldstein et al., 2013; Law et al., 2010; Lebreton and Borrero, 2013; Maximenko et al., 2012). In our study, we have found lower abundances of AMD in Rapa Nui and Juan Fernandez compared to the continental coast. Nevertheless, previous reports have found high densities of small-sized plastic AMD on Rapa Nui and in surrounding waters (Eriksen et al., 2013; Hidalgo-Ruz and Thiel, 2013; Ory et al., 2017). Small-sized plastics can be a result of breakdown of larger items that comprise an important fraction of AMD in oceanic gyres and on island beaches (Eriksen et al., 2014; Lusher, 2015; Miranda-Urbina et al., 2015). Nevertheless, since small-sized plastics are rarely sampled in typical AMD samplings, they are commonly underrepresented in many beach surveys. Therefore, it can be useful to apply different sampling methods according to micro-, meso- and macro-AMD (Lee et al., 2013) at these oceanic locations.

Results from our study indicated that AMD abundances are higher at the upper stations of the beaches, where AMD is distributed between the high tide lines and the supralittoral zone of beaches near frontal dunes, similar as in many other studies (Claereboudt, 2004; Costa et al., 2011; Galgani et al., 2015; Oigman-Pszczol and Creed, 2007; Silva-

Iñiguez and Fischer, 2003; Williams and Tudor, 2001). This could be indicative of direct deposition of AMD by beach users, which mostly rest in those sections of a beach (e.g. Bravo et al., 2009). However, this distribution pattern might also depend on environmental factors. For example, winter storms and wind might transport AMD further up the shore, into the dunes (Thornton and Jackson, 1998) or else, storm waves might pull AMD back to the sea (Ivar do Sul et al., 2011).

Beach clean-up services can also affect AMD abundances. In coastal areas of Brazil the beaches closest to highly populated areas had the highest amounts of AMD, but this pattern was disrupted on beaches with an efficient public cleaning system (Leite et al., 2014). The central area of Chile with a temperate climate and extensive sandy beaches receives high numbers of tourists year-round (González and Holtmann-Ahumada, 2017). Some of the most prioritized ecosystem services for tourists are beach and coastal scenery and their quietness (de Juan et al., 2017). Since AMD accumulation is a factor that jeopardizes beach scenery, extensive cleaning of tourist beaches is conducted in order to reduce AMD abundances, which might affect the patterns of AMD dynamics (Smith and Markic, 2013; Somerville et al., 2003).

4.3. *Temporal variation of AMD abundances*

No significant increase in AMD densities was observed for the 8 year period of our study, contrary to our prediction that the abundance of AMD at Chilean beaches would be increasing over time. Previous studies that have analyzed temporal changes in AMD densities are highly variable. Those studies cover wide spatial scales, from a single beach or island (e.g. Agustin et al., 2015; Walker, 1997) up to a continental scale (Ribic et al., 2011; Schulz et al., 2015b), and a wide range of temporal extensions, with a time frame of 2 to 25 years (e.g. Ribic, 1998; Schulz et al., 2015a). This high variability among studies limits the comparability among many studies (Browne et al., 2015).

A closer examination of temporal changes of AMD in studies covering similar time periods as our study (6 to 22 years) reveals that there are indeed no clear temporal trends in AMD abundances (Table 4). However, it is interesting to note that in the country with the lowest Human Development Index (HDI) (UNDP, 2016) and Education Index (EI) (UNDP, 2013) there is a tendency of AMD abundances to increase, while in the country with the highest HDI and EI, there is a tendency of decreasing litter abundances (Table 4). In the first case, an increasing abundance of AMD from 1994 to 2003 was reported at Rio Grande, Brazil (Tourinho and Fillmann, 2011), the country with the lowest HDI and EI. The authors suggested that the causes for increasing AMD abundances are related to socio-economic aspects, such as development of lifestyle, population growth, tourism activity and fishery activity. On the other hand, a steady decrease of AMD abundance from 1991 to 1999 was described for Anxious Bay, Australia (Edyvane et al., 2004), the country with the highest HDI and EI. In this case AMD items were related particularly to fishing activity, such as bait straps, baskets, and buoys. During the time of the study, some fisheries showed a marked reductions in fishing effort in the region, which led to a gradual decline of this sort of AMD. However, in the same study "a sharp increase in litter was recorded in 2000" and this was "probably due to stronger than average onshore surface flow (or Ekman Transport) in the western Eyre Peninsula and Bight region" (Edyvane et al., 2004).

Although these two examples are relatively consistent with our initial hypothesis, the majority of these studies did not conform to a consistent pattern. In some cases, an increasing trend was found only for certain AMD items, such as plastics and fishing nets (Nelms et al., 2016; Ribic et al., 2010), or a particular decreasing trend of AMD accumulation of fishing-related items (Schulz et al., 2015b). Moreover, at some single locations, analyses of AMD sources revealed contrasting temporal trends depending on their origins. For example, for the Mid-Atlantic US coasts Ribic et al. (2010) revealed that land-based and general-sources debris increased between the years 1997 to 2007, but ocean-based debris decreased, suggesting changes in human activities

Table 4
Studies that report temporal variations of AMD abundance, with their locations and time frame. HDI (Human Development Index) (UNDP, 2016) and Education Index (Mean and Expected Years of Schooling) (UNDP, 2013) were used as proxies for socio-economic growth and educational conditions of each country. Percentage of AMD variation was calculated by the difference between AMD abundance of the first and the last year of each study. Empty cells indicate no available data from the respective study.

Reference	Location	Time frame	HDI	Education index	General temporal AMD trend	% of AMD variation	Maximun of AMD	Mean of AMD	Units
Tourinho and Fillmann, 2011	Cassino beach, Brazil	1995–2006	0.755	0.661	Increasing	44%	10.7		Items m^{-1}
Ribic et al., 2011	US Caribbean	1997–2002	0.920	0.890	Decreasing	− 80%		193.4	Items 500 m^{-1}
Ribic et al., 2011	Eastern Gulf of Mexico	1997–2002	0.920	0.890	Decreasing	− 31%		38.1	Items 500 m^{-1}
Ribic et al., 2011	Western Gulf of Mexico	1997–2002	0.920	0.890	None			158.2	Items 500 m^{-1}
Our study	Chile	2008–2016	0.832	0.746	None		13.0	1.7	Items m^{-1}
Gago et al., 2014	Galicia, Spain	2001–2010	0.876	0.794	Increasing	70%	1785.0	1016	Items 100 m^{-1}
Nelms et al., 2016	UK	2005–2014	0.907	0.860	Increasing of some items	2.3 fold	0.3	0.009	Items m^{-1} $person^{-1}$
Agustin et al., 2015	Tern Island, Hawaii	1990–2012	0.915	0.889	None		40.0	15	Items day^{-1}
Morishige et al., 2007	French Frigate Shoals, Hawai'i	1990–2006	0.915	0.889	None			3085	Items $year^{-1}$
Ribic et al., 2012	North Pacific Coast	1998–2007	0.915	0.889	Decreasing	− 73%		28.2	Items 500 m^{-1}
Ribic et al., 2012	Southern California Bight	1998–2007	0.915	0.889	Decreasing	5–78%		69.5	Items 500 m^{-1}
Ribic et al., 2012	Hawai'i	1998–2007	0.915	0.889	Decreasing	40–65%		133.8	Items 500 m^{-1}
Ribic et al., 2010	Northeast Atlantic	1997–2007	0.915	0.889	None			51.2	Items 500 m^{-1}
Ribic et al., 2010	Mid-Atlantic	1997–2007	0.915	0.889	Land-based and general-sources increasing, but, ocean-based decreased			214.3	Items 500 m^{-1}
Ribic et al., 2010	Southeast Atlantic	1997–2007	0.915	0.889	Ocean-based decrease			41.6	Items 500 m^{-1}
Schulz et al., 2015b	North Sea, Europe	2002–2008	0.916	0.884	No general trend, but increasing at one site			217.5	Items 100 m^{-1}
Edyvane et al., 2004	Anxious Bay, Australia	1991–2000	0.935	0.926	Gradually declining from 1991 to 1999, but at 2000 suddenly increase	− 85%	15.0	3	kg km^{-1}

during that time period. Educational levels may also affect littering behavior of people (e.g. Eastman et al., 2013), and consequently AMD densities. Also, in some cases AMD variations were associated with oceanographic seasonal patterns, such as El Niño events (Gago et al., 2014; Morishige et al., 2007), and wind-induced drift and wave action (Agustin et al., 2015). These records show that AMD abundances are highly variable over time and are strongly affected by the local context of each study, particularly by a combination of social and oceanographic factors.

Thus, the expected link between economic development and pro-environmental behaviors (e.g. Morren and Grinstein, 2016) might not closely apply to AMD abundances on beaches, most likely because external factors (hydrodynamics, climate) are frequently disrupting this relationship. Additionally, the temporal extension and resolution of most currently available studies might still be too limited to determine the temporal trends in AMD abundances. Therefore, in order to rigorously examine this putative relationship between the amounts of marine litter and socio-economic development of a country (or region), future studies require long-term data sets, and also a better indicator of the littering and recycling habits of the local population.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.marpolbul.2017.11.014.

References

Aguilera, M.A., Broitman, B.R., Thiel, M., 2016. Artificial breakwaters as garbage bins: structural complexity enhances anthropogenic litter accumulation in marine intertidal habitats. Environ. Pollut. 214, 737–747. http://dx.doi.org/10.1016/j.envpol.2016.04.058.

Agustin, A.E., Merrifield, M.A., Potemra, J.T., Morishige, C., 2015. Temporal variability of marine debris deposition at Tern Island in the Northwestern Hawaiian Islands. Mar. Pollut. Bull. 101, 200–207. http://dx.doi.org/10.1016/j.marpolbul.2015.10.076.

Amaral-Zettler, L.A., Zettler, E.R., Slikas, B., Boyd, G.D., Melvin, D.W., Morrall, C.E., Proskurowski, G., Mincer, T.J., 2015. The biogeography of the Plastisphere: implications for policy. Front. Ecol. Environ. 13, 541–546. http://dx.doi.org/10.1890/150017.

Andrades, R., Martins, A.S., Fardim, L.M., Ferreira, J.S., Santos, R.G., 2016. Origin of marine debris is related to disposable packs of ultra-processed food. Mar. Pollut. Bull. 109, 192–195. http://dx.doi.org/10.1016/j.marpolbul.2016.05.083.

Araújo, M.C., Costa, M., 2007. An analysis of the riverine contribution to the solid wastes contamination of an isolated beach at the Brazilian Northeast. Manag. Environ. Qual. Int. J. 18, 6–12.

Astudillo, J., Bravo, M., Dumont, C., Thiel, M., 2009. Detached aquaculture buoys in the SE Pacific: potential dispersal vehicles for associated organisms. Aquat. Biol. 5, 219–231. http://dx.doi.org/10.3354/ab00151.

Barnes, D.K.A., Galgani, F., Thompson, R.C., Barlaz, M., 2009. Accumulation and fragmentation of plastic debris in global environments. Philos. Trans. R. Soc. Lond. Ser. B Biol. Sci. 364, 1985–1998. http://dx.doi.org/10.1098/rstb.2008.0205.

Benton, T.G., 1995. From castaways to throwaways: marine litter in the Pitcairn Islands. Biol. J. Linn. Soc. 56, 415–422. http://dx.doi.org/10.1006/bijl.1995.0077.

Blickley, L.C., Currie, J.J., Kaufman, G.D., 2016. Trends and drivers of debris accumulation on Maui shorelines: implications for local mitigation strategies. Mar. Pollut. Bull. 105, 292–298. http://dx.doi.org/10.1016/j.marpolbul.2016.02.007.

Borja, A., Galparsoro, I., Irigoien, X., Iriondo, A., Menchaca, I., Muxika, I., Pascual, M., Quincoces, I., Revilla, M., Germán Rodríguez, J., Santurtún, M., Solaun, O., Uriarte, A., Valencia, V., Zorita, I., 2011. Implementation of the European marine strategy framework directive: a methodological approach for the assessment of environmental status, from the Basque Country (Bay of Biscay). Mar. Pollut. Bull. 62, 889–904. http://dx.doi.org/10.1016/j.marpolbul.2011.03.031.

Bowman, D., Manor-Samsonov, N., Golikut, A., 1998. Dynamics of litter pollution on Israeli Mediterranean beaches: a budgetary, litter flux approach. J. Coast. Res. 14, 418–432. http://www.jstor.org/stable/4298796.

Bravo, M., Gallardo, M.D., Luna-Jorquera, G., Núñez, P., Vásquez, N., Thiel, M., 2009. Anthropogenic debris on beaches in the SE Pacific (Chile): results from a national survey supported by volunteers. Mar. Pollut. Bull. 58, 1718–1726. http://dx.doi.org/10.1016/j.marpolbul.2009.06.017.

Browne, M.A., Galloway, T.S., Thompson, R.C., 2010. Spatial patterns of plastic debris along estuarine shorelines. Environ. Sci. Technol. 44, 3404–3409. http://dx.doi.org/10.1021/es903784e.

Browne, M.A., Chapman, M.G., Thompson, R.C., Amaral-Zettler, L.A., Jambeck, J., Mallos, N.J., 2015. Spatial and temporal patterns of stranded intertidal marine debris: is there a picture of global change? Environ. Sci. Technol. 49, 7082–7094. http://dx.doi.org/10.1021/es5060572.

Cheung, P.K., Cheung, L.T.O., Fok, L., 2016. Seasonal variation in the abundance of marine plastic debris in the estuary of a subtropical macro-scale drainage basin in South China. Sci. Total Environ. 562, 658–665. http://dx.doi.org/10.1016/j.scitotenv.2016.04.048.

Claereboudt, M.R., 2004. Shore litter along sandy beaches of the Gulf of Oman. Mar. Pollut. Bull. 49, 770–777. http://dx.doi.org/10.1016/j.marpolbul.2004.06.004.

Corbin, C.J., Singh, J.G., 1993. Marine debris contamination of beaches in St. Lucia and Dominica. Mar. Pollut. Bull. 26, 325–328. http://dx.doi.org/10.1016/0025-326X(93)90575-5.

Costa, M.F., Silva-Cavalcanti, J.S., Barbosa, C.C., Portugal, J.L., Barletta, M., 2011. Plastics buried in the inter-tidal plain of a tropical estuarine ecosystem. J. Coast. Res. SI 64, 339–343.

de Juan, S., Gelcich, S., Fernandez, M., 2017. Integrating stakeholder perceptions and preferences on ecosystem services in the management of coastal areas. Ocean Coast. Manag. 136, 38–48. http://dx.doi.org/10.1016/j.ocecoaman.2016.11.019.

Eastman, L.B., Núñez, P., Crettier, B., Thiel, M., 2013. Identification of self-reported user behavior, education level, and preferences to reduce littering on beaches–a survey from the SE Pacific. Ocean Coast. Manag. 78, 18–24.

Eastman, L., Hidalgo-Ruz, V., Macaya, V., Nuñez, P., Thiel, M., 2014. The potential for young citizen scientist projects: a case study of Chilean schoolchildren collecting data on marine litter. JICZM 14, 569–579. http://dx.doi.org/10.5894/rgci507.

Edyvane, K.S., Dalgetty, A., Hone, P.W., Higham, J.S., Wace, N.M., 2004. Long-term marine litter monitoring in the remote Great Australian Bight, South Australia. Mar. Pollut. Bull. 48, 1060–1075. http://dx.doi.org/10.1016/j.marpolbul.2003.12.012.

Eriksen, M., Maximenko, N., Thiel, M., Cummins, A., Lattin, G., Wilson, S., Hafner, J., Zellers, A., Rifman, S., 2013. Plastic pollution in the South Pacific subtropical gyre. Mar. Pollut. Bull. 68, 71–76. http://dx.doi.org/10.1016/j.marpolbul.2012.12.021.

Eriksen, M., Lebreton, L.C., Carson, H.S., Thiel, M., Moore, C.J., Borerro, J.C., Galgani, F., Ryan, P.G., Reisser, J., 2014. Plastic pollution in the world's oceans: more than 5 trillion plastic pieces weighing over 250,000 tons afloat at sea. PLoS One 9 (12), e111913. http://dx.doi.org/10.1371/journal.pone.0111913.

Eriksson, C., Burton, H., Fitch, S., Schulz, M., van den Hoff, J., 2013. Daily accumulation rates of marine debris on sub-Antarctic island beaches. Mar. Pollut. Bull. 66, 199–208. http://dx.doi.org/10.1016/j.marpolbul.2012.08.026.

Gago, J., Lahuerta, F., Antelo, P., 2014. Characteristics (abundance, type and origin) of beach litter on the Galician coast (NW Spain) from 2001 to 2010. Sci. Mar. 78, 125–134. http://dx.doi.org/10.3989/scimar.03883.31B.

Galgani, F., Hanke, G., Maes, T., 2015. Global distribution, composition and abundance of marine litter. In: Bergmann, M., Gutow, L., Klages, M. (Eds.), Marine Anthropogenic Litter. Springer Verlag, Berlin, pp. 29–56. http://dx.doi.org/10.1007/978-3-319-16510-3.

Goldstein, M.C., Titmus, A.J., Ford, M., 2013. Scales of spatial heterogeneity of plastic marine debris in the northeast Pacific Ocean. PLoS One 8 (11), e80020. http://dx.doi.org/10.1371/journal.pone.0080020.

González, S.A., Holtmann-Ahumada, G., 2017. Quality of tourist beaches of northern Chile: a first approach for ecosystem-based management. Ocean Coast. Manag. 137, 154–164. http://dx.doi.org/10.1016/j.ocecoaman.2016.12.022.

Goto, T., Shibata, H., 2015. Changes in abundance and composition of anthropogenic marine debris on the continental slope off the Pacific coast of northern Japan, after the March 2011 Tohoku earthquake. Mar. Pollut. Bull. 95, 234–241. http://dx.doi.org/10.1016/j.marpolbul.2015.04.011.

Green, D.S., Boots, B., Blockley, D.J., Rocha, C., Thompson, R., 2015. Impacts of discarded plastic bags on marine assemblages and ecosystem functioning. Environ. Sci. Technol. 49, 5380–5389. http://dx.doi.org/10.1021/acs.est.5b00277.

Gregory, M.R., 2009. Environmental implications of plastic debris in marine settings–entanglement, ingestion, smothering, hangers-on, hitch-hiking and alien invasions. Philos. Trans. R. Soc. Lond. Ser. B Biol. Sci. 364, 2013–2025. http://dx.doi.org/10.1098/rstb.2008.0265.

Hidalgo-Ruz, V., Thiel, M., 2013. Distribution and abundance of small plastic debris on beaches in the SE Pacific (Chile): a study supported by a citizen science project. Mar. Environ. Res. 87–88, 12–18. http://dx.doi.org/10.1016/j.marenvres.2013.02.015.

Hinojosa, I.A., Thiel, M., 2009. Floating marine debris in fjords, gulfs and channels of Southern Chile. Mar. Pollut. Bull. 58, 341–350. http://dx.doi.org/10.1016/j.marpolbul.2008.10.020.

Hong, S., Lee, J., Kang, D., Choi, H.W., Ko, S.H., 2014. Quantities, composition, and sources of beach debris in Korea from the results of nationwide monitoring. Mar. Pollut. Bull. 84, 27–34. http://dx.doi.org/10.1016/j.marpolbul.2014.05.051.

Ivar do Sul, J.A., Costa, M.F., 2007. Marine debris review for Latin America and the wider Caribbean region: from the 1970s until now, and where do we go from here? Mar. Pollut. Bull. 54, 1087–1104. http://dx.doi.org/10.1016/j.marpolbul.2007.05.004.

Ivar do Sul, J.A., Spengler, A., Costa, M.F., 2009. Here, there and everywhere. Small plastic fragments and pellets on beaches of Fernando de Noronha (Equatorial Western Atlantic). Mar. Pollut. Bull. 58, 1236–1238. http://dx.doi.org/10.1016/j.marpolbul.2009.05.004.

Ivar do Sul, J.A., Santos, I.R., Friedrich, A.C., Matthiensen, A., Fillmann, G., 2011. Plastic pollution at a sea turtle conservation area in NE Brazil: contrasting developed and undeveloped beaches. Estuar. Coasts 34, 814–823. http://dx.doi.org/10.1007/s12237-011-9392-8.

Jambeck, J.R., Geyer, R., Wilcox, C., Siegler, T.R., Perryman, M., Andrady, A., Narayan, R., Law, K.L., 2015. Plastic waste inputs from land into the ocean. Science 347, 768–771. http://dx.doi.org/10.1126/science.1260352.

Jang, Y.C., Lee, J., Hong, S., Lee, J.S., Shim, W.J., Song, Y.K., 2014. Sources of plastic marine debris on beaches of Korea: more from the ocean than the land. Ocean Sci. J. 49, 151–162. http://dx.doi.org/10.1007/s12601-014-0015-8.

Khordagui, H., Abu-Hilal, A., 1994. Industrial plastic on the southern beaches of the Arabian Gulf and the western beaches of the Gulf of Oman. Environ. Pollut. 84, 325–327. http://dx.doi.org/10.1016/0269-7491(94)90143-0.

Kiessling, T., Gutow, L., Thiel, M., 2015. Marine litter as habitat and dispersal vector. In: Bergmann, M., Gutow, L., Klages, M. (Eds.), Marine Anthropogenic Litter. Springer Verlag, Berlin, pp. 141–181. http://dx.doi.org/10.1007/978-3-319-16510-3.

Kiessling, T., Salas, S., Mutafoglu, K., Thiel, M., 2017. Who cares about dirty beaches? Evaluating environmental awareness and action on coastal litter in Chile. Ocean Coast. Manag. 137, 82–95. http://dx.doi.org/10.1016/j.ocecoaman.2016.11.029.

Kühn, S., Rebolledo, E.L.B., van Franeker, J.A., 2015. Deleterious effects of litter on marine life. In: Bergmann, M., Gutow, L., Klages, M. (Eds.), Marine Anthropogenic Litter. Springer Verlag, Berlin, pp. 75–116. http://dx.doi.org/10.1007/978-3-319-16510-3.

Kuo, F.J., Huang, H.W., 2014. Strategy for mitigation of marine debris: analysis of sources and composition of marine debris in northern Taiwan. Mar. Pollut. Bull. 83, 70–78. http://dx.doi.org/10.1016/j.marpolbul.2014.04.019.

Law, K.L., Morét-Ferguson, S., Maximenko, N.A., Proskurowski, G., Peacock, E.E., Hafner, J., Reddy, C.M., 2010. Plastic accumulation in the North Atlantic subtropical gyre. Science 329, 1185–1188. http://dx.doi.org/10.1126/science.1192321.

Lebreton, L.C.M., Borrero, J.C., 2013. Modeling the transport and accumulation floating debris generated by the 11 march 2011 Tohoku tsunami. Mar. Pollut. Bull. 66, 53–58. http://dx.doi.org/10.1016/j.marpolbul.2012.11.013.

Lee, J., Hong, S., Song, Y.K., Hong, S.H., Jang, Y.C., Jang, M., Heo, N.W., Han, G.M., Lee, M.J., Kang, D., Shim, W.J., 2013. Relationships among the abundances of plastic debris in different size classes on beaches in South Korea. Mar. Pollut. Bull. 77, 349–354. http://dx.doi.org/10.1016/j.marpolbul.2013.08.013.

Leite, A.S., Santos, L.L., Costa, Y., Hatje, V., 2014. Influence of proximity to an urban center in the pattern of contamination by marine debris. Mar. Pollut. Bull. 81, 242–247. http://dx.doi.org/10.1016/j.marpolbul.2014.01.032.

Li, W.C., Tse, H.F., Fok, L., 2016. Plastic waste in the marine environment: a review of sources, occurrence and effects. Sci. Total Environ. 566–567, 333–349. http://dx.doi.org/10.1016/j.scitotenv.2016.05.084.

Liu, T.-K., Wang, M.-W., Chen, P., 2013. Influence of waste management policy on the characteristics of beach litter in Kaohsiung, Taiwan. Mar. Pollut. Bull. 72, 99–106. http://dx.doi.org/10.1016/j.marpolbul.2013.04.015.

Lucas, Z., 1992. Monitoring persistent litter in the marine environment on Sable Island, Nova Scotia. Mar. Pollut. Bull. 24, 192–199. http://dx.doi.org/10.1016/0025-326X(92)90529-F.

Lusher, A., 2015. Microplastics in the marine environment: distribution, interactions and effects. In: Bergmann, M., Gutow, L., Klages, M. (Eds.), Marine Anthropogenic Litter. Springer Verlag, Berlin, pp. 245–307. http://dx.doi.org/10.1007/978-3-319-16510-3.

Madzena, A., Lasiak, T., 1997. Spatial and temporal variations in beach litter on the Transkei coast of South Africa. Mar. Pollut. Bull. 34, 900–907. http://dx.doi.org/10.1016/S0025-326X(97)00052-0.

Martinez, E., Maamaatuaiahutapu, K., Taillandier, V., 2009. Floating marine debris surface drift: convergence and accumulation toward the South Pacific subtropical gyre. Mar. Pollut. Bull. 58, 1347–1355. http://dx.doi.org/10.1016/j.marpolbul.2009.04.022.

Maximenko, N., Hafner, J., Niiler, P., 2012. Pathways of marine debris derived from trajectories of Lagrangian drifters. Mar. Pollut. Bull. 65, 51–62. http://dx.doi.org/10.1016/j.marpolbul.2011.04.016.

Miloslavich, P., Klein, E., Díaz, J.M., Hernández, C.E., Bigatti, G., Campos, L., Artigas, F., Castillo, J., Penchaszadeh, P.E., Neill, P.E., Carranza, A., Retana, M.V., Díaz de Astarloa, J.M., Lewis, M., Yorio, P., Piriz, M.L., Rodríguez, D., Valentin, Y.Y., Gamboa, L., Martín, A., 2011. Marine biodiversity in the Atlantic and Pacific coasts of South America: knowledge and gaps. PLoS One 6, e14631. http://dx.doi.org/10.1371/journal.pone.0014631.

Miranda-Urbina, D., Thiel, M., Luna-Jorquera, G., 2015. Litter and seabirds found across a longitudinal gradient in the South Pacific Ocean. Mar. Pollut. Bull. 96, 235–244. http://dx.doi.org/10.1016/j.marpolbul.2015.05.021.

Moore, C.J., 2008. Synthetic polymers in the marine environment: a rapidly increasing, long-term threat. Environ. Res. 108, 131–139. http://dx.doi.org/10.1016/j.envres.2008.07.025.

Moreira, F.T., Prantoni, A.L., Martini, B., de Abreu, M.A., Stoiev, S.B., Turra, A., 2015. Small-scale temporal and spatial variability in the abundance of plastic pellets on sandy beaches: methodological considerations for estimating the input of microplastics. Mar. Pollut. Bull. 102, 114–121. http://dx.doi.org/10.1016/j.marpolbul.2015.11.051.

Moriarty, M., Pedreschi, D., Stokes, D., Dransfeld, L., Reid, D.G., 2016. Spatial and temporal analysis of litter in the Celtic Sea from groundfish survey data: lessons for monitoring. Mar. Pollut. Bull. 103, 195–205. http://dx.doi.org/10.1016/j.marpolbul.2015.12.019.

Morishige, C., Donohue, M.J., Flint, E., Swenson, C., Woolaway, C., 2007. Factors affecting marine debris deposition at French frigate shoals, northwestern Hawaiian islands marine National Monument, 1990-2006. Mar. Pollut. Bull. 54, 1162–1169. http://dx.doi.org/10.1016/j.marpolbul.2007.04.014.

Morren, M., Grinstein, A., 2016. Explaining environmental behavior across borders: a meta-analysis. J. Environ. Psychol. 47, 91–106. http://dx.doi.org/10.1016/j.jenvp.2016.05.003.

Nelms, S., Coombes, C., Foster, L., Galloway, T., Godley, B., Lindeque, P., Witt, M., 2016. Marine anthropogenic litter on British beaches: a 10-year nationwide assessment using citizen science data. Sci. Total Environ. 579, 1399–1409. http://dx.doi.org/10.1016/j.scitotenv.2016.11.137.

Newman, S., Watkins, E., Farmer, A., ten Brink, P., Schweitzer, J.-P., 2015. The economics of marine litter. In: Bergmann, M., Gutow, L., Klages, M. (Eds.), Marine Anthropogenic Litter. Springer Verlag, Berlin, pp. 367–394. http://dx.doi.org/10.1007/978-3-319-16510-3.

Oigman-Pszczol, S.S., Creed, J.C., 2007. Quantification and classification of marine litter on beaches along Armação dos Búzios, Rio de Janeiro, Brazil. J. Coast. Res. 232, 421–428. http://dx.doi.org/10.2112/1551-5036(2007)23[421:QACOML]2.0.CO;2.

Ory, N.C., Sobral, P., Ferreira, J.L., Thiel, M., 2017. Amberstripe scad *Decapterus muroadsi* (Carangidae) fish ingest blue microplastics resembling their copepod prey along the coast of Rapa Nui (Easter Island) in the South Pacific subtropical gyre. Sci. Total Environ. 586, 430–437. http://dx.doi.org/10.1016/j.scitotenv.2017.01.175.

Portz, L., Manzolli, R.P., Do Sul, J.a.I., 2011. Marine debris on Rio Grande do Sul north coast, Brazil: spatial and temporal patterns. JICZM 11, 41–48. http://dx.doi.org/10.5894/rgci187.

R Development Core Team, 2008. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria Available online at. http://www.R-project.org/.

Rech, S., Macaya-Caquilpán, V., Pantoja, J.F., Rivadeneira, M.M., Jofre Madariaga, D., Thiel, M., 2014. Rivers as a source of marine litter - a study from the SE Pacific. Mar. Pollut. Bull. 82, 66–75. http://dx.doi.org/10.1016/j.marpolbul.2014.03.019.

Ribic, C.A., 1998. Use of indicator items to monitor marine debris on a New Jersey beach from 1991 to 1996. Mar. Pollut. Bull. 36, 887–891. http://dx.doi.org/10.1016/S0025-326X(98)00064-2.

Ribic, C.A., Sheavly, S.B., Rugg, D.J., Erdmann, E.S., 2010. Trends and drivers of marine debris on the Atlantic coast of the United States 1997–2007. Mar. Pollut. Bull. 60, 1231–1242. http://dx.doi.org/10.1016/j.marpolbul.2010.03.021.

Ribic, C.A., Sheavly, S.B., Rugg, D.J., 2011. Trends in marine debris in the U.S. Caribbean and the Gulf of Mexico 1996–2003. JICZM 11, 7–19. http://dx.doi.org/10.5894/rgci181.

Ribic, C.A., Sheavly, S.B., Rugg, D.J., Erdmann, E.S., 2012. Trends in marine debris along the U.S. Pacific coast and Hawai'i 1998-2007. Mar. Pollut. Bull. 64, 994–1004. http://dx.doi.org/10.1016/j.marpolbul.2012.02.008.

Rochman, C.M., Hoh, E., Kurobe, T., Teh, S.J., 2013. Ingested plastic transfers hazardous chemicals to fish and induces hepatic stress. Sci. Rep. 3, 3263. http://dx.doi.org/10.1038/srep03263.

Rosevelt, C., Los Huertos, M., Garza, C., Nevins, H.M., 2013. Marine debris in central California: quantifying type and abundance of beach litter in Monterey Bay, CA. Mar. Pollut. Bull. 71, 299–306. http://dx.doi.org/10.1016/j.marpolbul.2013.01.015.

Sadri, S.S., Thompson, R.C., 2014. On the quantity and composition of floating plastic debris entering and leaving the Tamar estuary, Southwest England. Mar. Pollut. Bull. 81, 55–60. http://dx.doi.org/10.1016/j.marpolbul.2014.02.020.

Santos, I.R., Friedrich, A.C., Wallner-Kersanach, M., Fillmann, G., 2005. Influence of socio-economic characteristics of beach users on litter generation. Ocean Coast. Manag. 48, 742–752. http://dx.doi.org/10.1016/j.ocecoaman.2005.08.006.

Santos, I.R., Friedrich, A.C., Ivar do Sul, J.A., 2009. Marine debris contamination along undeveloped tropical beaches from northeast Brazil. Environ. Monit. Assess. 148, 455–462. http://dx.doi.org/10.1007/s10661-008-0175-z.

Schulz, M., Clemens, T., Förster, H., Harder, T., Fleet, D., Gaus, S., Grave, C., Flegel, I., Schrey, E., Hartwig, E., 2015a. Statistical analyses of the results of 25 years of beach litter surveys on the south-eastern North Sea coast. Mar. Environ. Res. 109, 21–27. http://dx.doi.org/10.1016/j.marenvres.2015.04.007.

Schulz, M., Krone, R., Dederer, G., Wätjen, K., Matthies, M., 2015b. Comparative analysis of time series of marine litter surveyed on beaches and the seafloor in the south-eastern North Sea. Mar. Environ. Res. 106, 61–67. http://dx.doi.org/10.1016/j.marenvres.2015.03.005.

Sheavly, S.B., Register, K.M., 2007. Marine debris & plastics: environmental concerns, sources, impacts and solutions. J. Polym. Environ. 15, 301–305. http://dx.doi.org/10.1007/s10924-007-0074-3.

Silva-Iñiguez, L., Fischer, D.W., 2003. Quantification and classification of marine litter on the municipal beach of Ensenada, Baja California, Mexico. Mar. Pollut. Bull. 46, 132–138. http://dx.doi.org/10.1016/S0025-326X(02)00216-3.

Smith, S.D.A., Markic, A., 2013. Estimates of marine debris accumulation on beaches are strongly affected by the temporal scale of sampling. PLoS One 8, 8–13. http://dx.doi.org/10.1371/journal.pone.0083694.

Somerville, S.E., Miller, K.L., Mair, J.M., 2003. Assessment of the aesthetic quality of a selection of beaches in the firth of forth, Scotland. Mar. Pollut. Bull. 46, 1184–1190. http://dx.doi.org/10.1016/S0025-326X(03)00126-7.

Storrier, K.L., McGlashan, D.J., 2006. Development and management of a coastal litter campaign: the voluntary coastal partnership approach. Mar. Policy 30, 189–196. http://dx.doi.org/10.1016/j.marpol.2005.01.002.

Storrier, K.L., McGlashan, D.J., Bonellie, S., Velander, K., 2007. Beach litter deposition at a selection of beaches in the Firth of Forth, Scotland. J. Coast. Res. 23, 813–822. http://dx.doi.org/10.2112/04-0251.1.

Thiel, M., Gutow, L., 2005. The ecology of rafting in the marine environment. I. The floating substrata. Oceanogr. Mar. Biol. Annu. Rev. 42, 181–264. http://dx.doi.org/10.1201/9780203507810.ch6.

Thiel, M., Hinojosa, I.A., Miranda, L., Pantoja, J.F., Rivadeneira, M.M., Vásquez, N., 2013. Anthropogenic marine debris in the coastal environment: a multi-year comparison between coastal waters and local shores. Mar. Pollut. Bull. 71, 307–316. http://dx.doi.org/10.1016/j.marpolbul.2013.01.005.

Thornton, L., Jackson, N., 1998. Spatial and temporal variations in debris accumulation and composition on an estuarine shoreline, Cliffwood Beach, New Jersey, USA. Mar. Pollut. Bull. 36, 705–711. http://dx.doi.org/10.1016/S0025-326X(98)00041-1.

Tourinho, P.S., Fillmann, G., 2011. Temporal trend of litter contamination at Cassino beach, Southern Brazil. JICZM 11, 97–102. http://dx.doi.org/10.5894/rgci201.

Turra, A., Manzano, A.B., Dias, R.J.S., Mahiques, M.M., Barbosa, L., Balthazar-Silva, D., Moreira, F.T., 2014. Three-dimensional distribution of plastic pellets in sandy beaches: shifting paradigms. Sci. Rep. 4, 4435. http://dx.doi.org/10.1038/srep04435.

UNDP, 2013. Education Index From Human Development Report Office [Online]. UNDP Available from. http://hdr.undp.org/es/content/education-index, Accessed date: April 2017.

UNDP, 2016. Human Development Report [Online]. UNDP Available: http://hdr.undp.org/sites/default/files/2016_human_development_report.pdf, Accessed date: April 2017.

Vegter, A.C., Barletta, M., Beck, C., Borrero, J., Burton, H., Campbell, M.L., Costa, M.F., Eriksen, M., Eriksson, C., Estrades, A., Gilardi, K.V.K., Hardesty, B.D., Ivar do Sul, J.A., Lavers, J.L., Lazar, B., Lebreton, L., Nichols, W.J., Ribic, C.A., Ryan, P.G., Schuyler, Q.A., Smith, S.D.A., Takada, H., Townsend, K.A., Wabnitz, C.C.C., Wilcox, C., Young, L.C., Hamann, M., 2014. Global research priorities to mitigate plastic pollution impacts on marine wildlife. Endanger. Species Res. 25, 225–247. http://dx.doi.org/10.3354/esr00623.

Veiga, J.M., Vlachogianni, T., Pahl, S., Thompson, R.C., Kopke, K., Doyle, T.K., Hartley, B.L., Maes, T., Orthodoxou, D.L., Loizidou, X.I., Alampei, I., 2016. Enhancing public awareness and promoting co-responsibility for marine litter in Europe: the challenge of MARLISCO. Mar. Pollut. Bull. 102, 309–315. http://dx.doi.org/10.1016/j.marpolbul.2016.01.031.

Velander, K.A., Mocogni, M., 1998. Maritime litter and sewage contamination at Cramond Beach Edinburgh - a comparative study. Mar. Pollut. Bull. 36, 385–389. http://dx.doi.org/10.1016/S0025-326X(97)00204-X.

Walker, T., 1997. Marine debris surveys at Bird Island, South Georgia 1990–1995. Mar. Pollut. Bull. 34, 61–65. http://dx.doi.org/10.1016/S0025-326X(96)00053-7.

Watters, D.L., Yoklavich, M.M., Love, M.S., Schroeder, D.M., 2010. Assessing marine debris in deep seafloor habitats off California. Mar. Pollut. Bull. 60, 131–138. http://dx.doi.org/10.1016/j.marpolbul.2009.08.019.

Williams, A.T., Tudor, D.T., 2001. Litter burial and exhumation: spatial and temporal distribution on a cobble pocket beach. Mar. Pollut. Bull. 42, 1031–1039.

Willoughby, N.G., Sangkoyo, H., Lakaseru, B.O., 1997. Beach litter: an increasing and changing problem for Indonesia. Mar. Pollut. Bull. 34, 469–478. http://dx.doi.org/10.1016/S0025-326X(96)00141-5.

Zuur, A.F., Ieno, E.N., Walker, N.J., Saveliev, A.A., Smith, G.M., 2009. Mixed Effects Models and Extensions in Ecology with R. Springer, New York, pp. 574. http://dx.doi.org/10.1007/978-0-387-87458-6.