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Artificial breakwaters as garbage bins: Structural complexity enhances anthropogenic litter accumulation in marine intertidal habitats☆

Moisés A. Aguilera [a, b, *], Bernardo R. Broitman [b], Martin Thiel [a, b]

[a] Departamento de Biología Marina, Facultad de Ciencias del Mar, Universidad Católica del Norte, Larrondo, 1281, Coquimbo, Chile
[b] Centro de Estudios Avanzados en Zonas Áridas (CEAZA), Ossandón 877, Coquimbo, Chile

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ABSTRACT

Coastal urban infrastructures are proliferating across the world, but knowledge about their emergent impacts is still limited. Here, we provide evidence that urban artificial reefs have a high potential to accumulate the diverse forms of litter originating from anthropogenic activities around cities. We test the hypothesis that the structural complexity of urban breakwaters, when compared with adjacent natural rocky intertidal habitats, is a driver of anthropogenic litter accumulation. We determined litter abundances at seven sites (cities) and estimated the structural complexity in both urban breakwaters and adjacent natural habitats from northern to central Chile, spanning a latitudinal gradient of ~15° (18°S to 33°S). Anthropogenic litter density was significantly higher in coastal breakwaters when compared to natural habitats (~15.1 items m^{-2} on artificial reefs versus 7.4 items m^{-2} in natural habitats) at all study sites, a pattern that was temporally persistent. Different litter categories were more abundant on the artificial reefs than in natural habitats, with local human population density and breakwater extension contributing to increase the probabilities of litter occurrence by ~10%. In addition, structural complexity was about two-fold higher on artificial reefs, with anthropogenic litter density being highest at intermediate levels of structural complexity. Therefore, the spatial structure characteristic of artificial reefs seems to enhance anthropogenic litter accumulation, also leading to higher residence time and degradation potential. Our study highlights the interaction between coastal urban habitat modification by establishment of artificial reefs, and pollution. This emergent phenomenon is an important issue to be considered in future management plans and the engineering of coastal ecosystems.

1. Introduction

Urbanization is one of the most critical and increasing drivers of species loss and ecosystem degradation worldwide (e.g. Bulleri and Chapman, 2010; Pickett et al., 2014; Vitousek et al., 1997). The extent to which urban expansion can impact adjacent ecosystems has been related to different anthropogenic drivers like population density, city area, economic/industrial status and cultural profiles (Pickett et al., 2014; Thompson, 2015). An important impact of cities on their surrounding ecosystems is that they export substantial amounts of multiple pollutants (Pickett et al., 2014) and anthropogenic litter (Wright et al., 2013; Thompson, 2015).

In coastal ecosystems the rate of coastal zone modification by industrialization is increasing (Gittman et al., 2015; He et al., 2014). In economically growing countries, the presence of coastal artificial habitats will increase in future decades following demands of coastal reclamation for economic activities and tourism revenues (He et al., 2014; Hill, 2015; Perkins et al., 2015), and the construction of infrastructures like ports and residential buildings (Airoldi et al., 2015; Bulleri and Chapman, 2010; Gittman et al., 2015).

Intertidal and subtidal reefs that were constructed in association with urban areas (artificial reefs) harbour fewer species when compared with natural adjacent ones (e.g. Firth et al., 2013, 2014). Coastal artificial reefs built of granite boulders can provide shelter for some species assemblages (e.g. fishes Burt et al., 2011, 2012)) which seems to be related to its large-scale (dozens of cm to meters) structural complexity. However, these habitats are expected to have low spatial heterogeneity when compared to natural habitats given the lack of biogenic microhabitats (Aguilera et al., 2014; Firth et al., 2014, 2013).

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* Corresponding author. Departamento de Biología Marina, Facultad de Ciencias del Mar, Universidad Católica del Norte, Larrondo, 1281, Coquimbo, Chile.
E-mail address: moises.aguilera@ceaza.cl (M.A. Aguilera).

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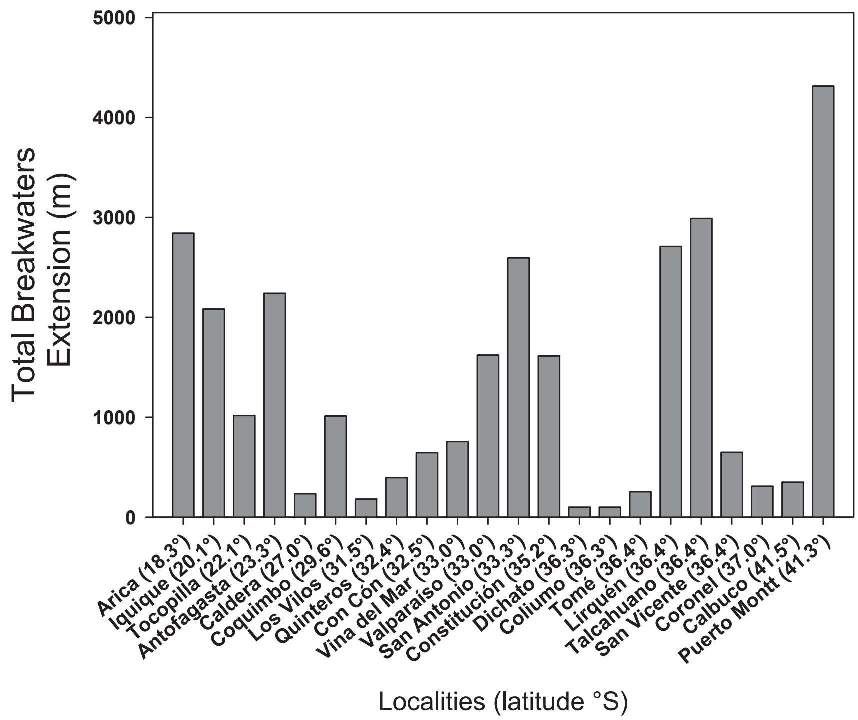

Fig. 1. Urban granite breakwaters linear length (meters) estimated at different localities across the Chilean coast (see Supporting information for details of the methodology). Latitude (°S) for each locality is shown in parenthesis.

Importantly, coastal habitat modification can 'open the door' to increased pollution by waste materials (e.g. plastic debris; Browne et al., 2015; Thompson, 2015). According to the specific spatial designs and anthropogenic uses of artificial reefs they may constitute hotspots for the accumulation of litter derived from human activities like recreational and commercial activities. Structural complexity of granite breakwaters created by spaces between boulders, or cavities among tetrapods, might enhance entrapment of anthropogenic litter (hereafter; AL), thereby favouring its accumulation. Commonly, anthropogenic litter accumulated on beaches cause a great risk to coastal biota (e.g. Browne et al., 2015; Thiel et al., 2011; Wright et al., 2013), with strong potential impacts on economy and human health (Sheavly and Register, 2007). However, the accumulation and residence time of AL in artificial intertidal reefs such as coastal breakwaters are unknown.

The South Eastern Pacific shores are polluted to varying degrees by anthropogenic litter (e.g. Bravo et al., 2009; Do Sul and da Costa, 2007). Studies of the composition of litter in coastal waters and on sandy beaches along the Chilean coast (see Bravo et al., 2009; Hinojosa and Thiel, 2009; Thiel et al., 2013) showed that most litter material had a local origin (Bravo et al., 2009; Hinojosa and Thiel, 2009; Thiel et al., 2013) when compared with other coasts where AL had more remote sources (e.g. Japan, Reimer et al., 2015). These studies have focused on AL accumulation along homogeneous natural shorelines, but little is known about the role of artificial infrastructures like breakwaters in AL accumulation. Economic and tourism development are the main drivers behind new coastal infrastructure (see for example; Bulleri and Chapman, 2010; Perkins et al., 2015), but protection against extreme natural events like storm surge and tsunamis are also important factors driving shoreline hardening (Gittman et al., 2015; Mimura et al., 2011). In this context, identifying the multiple impacts from urban infrastructures on coastal ecosystems can help managers and decision-makers to adopt more specific strategies for building and monitoring shoreline armouring (Scyphers et al., 2014).

As in other countries, shoreline hardening is expanding along the coast of Chile (Chilean Ministry of Public Work, MOP, and see Fig. 1). Most artificial structures are part of urban areas or incorporate public accesses that facilitate the use of shorelines for activities such as recreation or fishing. The purpose of this study was to examine if the abundance of AL is higher on urban coastal breakwaters than on adjacent, natural rocky platforms, and to determine if the structural complexity characteristics of both natural and artificial habitats influences AL distribution on them. We hypothesized that AL is more abundant on urban breakwaters compared with adjacent natural rocky platforms, following differences in patterns of structural complexity between habitats. Coastal human population density is commonly expected to contribute to an increase of AL accumulating in natural habitats nearby, with breakwater spatial extension potentially being directly related to the total accumulation of AL. Hence, we also examined the relationship between total AL density in breakwaters and local human population density and/or breakwater length.

2. Materials and methods

2.1. Details of the study region

The coast of Chile has an extension of about 6435 km, with more than 45% of the Chilean population living in the coastal zone (INE, 2012). Armouring of the coastal area, from 18°S to 41°S, is one of the most important management conflicts, and is proceeding either to expand human settlements, as disaster prevention/mitigation infrastructure or for the development of new activities related to tourism, deep-water harbours or leisure. Coastal cities in northern to central Chile (18°S to 33°S) develop economic activities related to tourism, and harbours are dedicated to fishing and shipping. Specifically, northern cities (18°–30°S) are located within the boundaries of the Atacama Desert characterized by low rain frequency

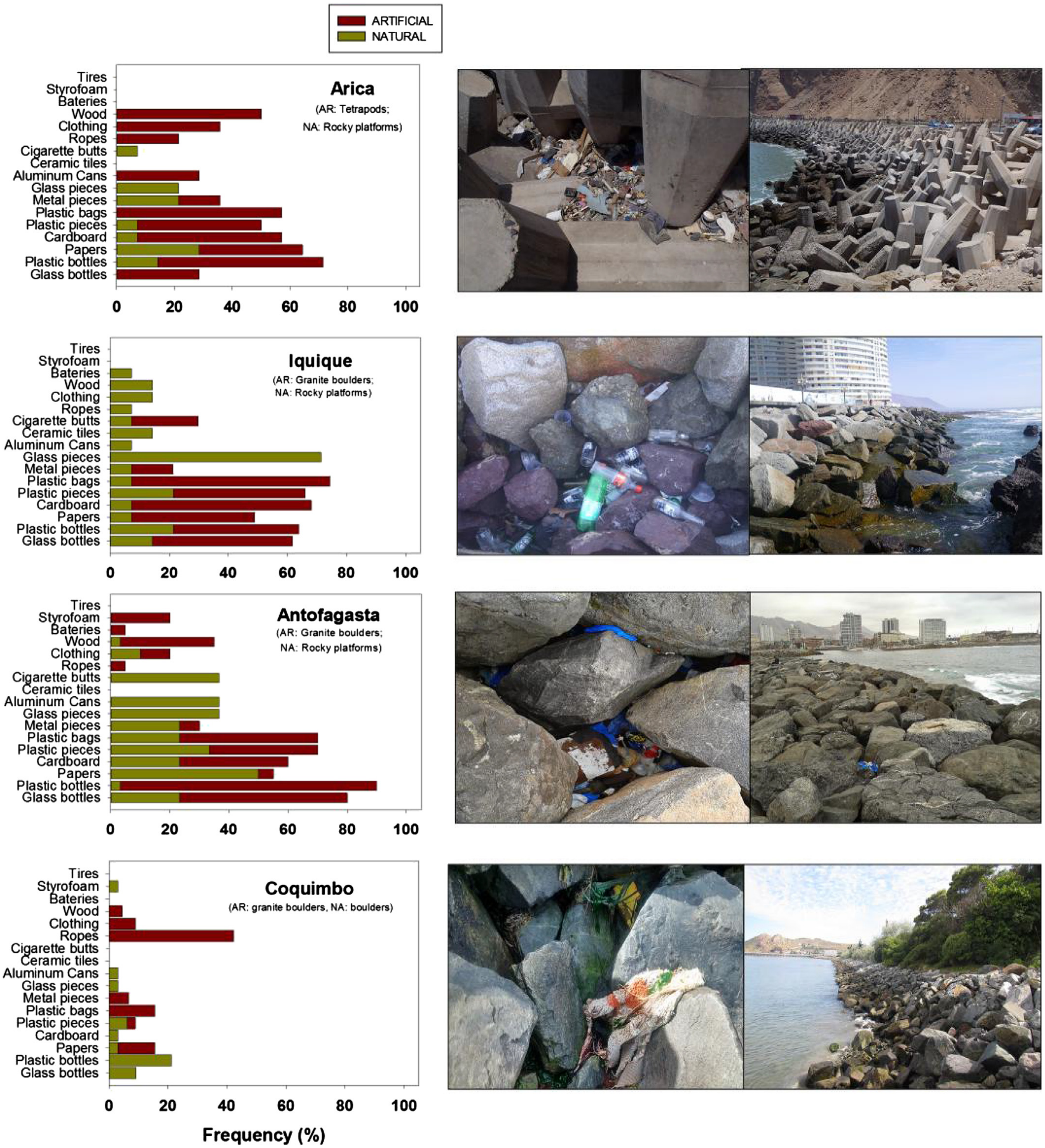

Fig. 2. Histograms of anthropogenic litter (AL) categories found at different localities and habitats (Artificial: breakwaters, Natural: rocky platforms) across the Chilean coast, and general view of different AL material observed on the sampled breakwaters at each locality. Sampling consisted of high shore transect with 35–45 0.25 m^{-2} quadrats positioned along each granite breakwater and adjacent (100 m–1 km far) natural rocky platforms at each locality. Characteristics of each sampled habitat are presented in each histogram: AR: artificial reefs and NA: natural habitat. "Styrofoam" is considered here as generic label for both; Expanded Polyestyrene (EPS) and Extruded Polyestyrene (XPS), which were pooled for analyses.

and few active rivers, increasing opportunities for living and recreational activities on the coast. Thus, the coastal cities in that range correspond to the most important urban centres in the region with higher human population pressure on marine ecosystems. Coastal cities present in central Chile (31°–34°S) correspond to the most populated coastal localities in the country (e.g. Valparaíso, Viña del Mar, INE., 2012), which harbour higher tourism and port activities. Coastal cities in central-southern Chile (35°S to 41°S), on the other hand, are mostly harbours related to forestry, fishing and aquaculture. Central to southern coastal cities are commonly influenced

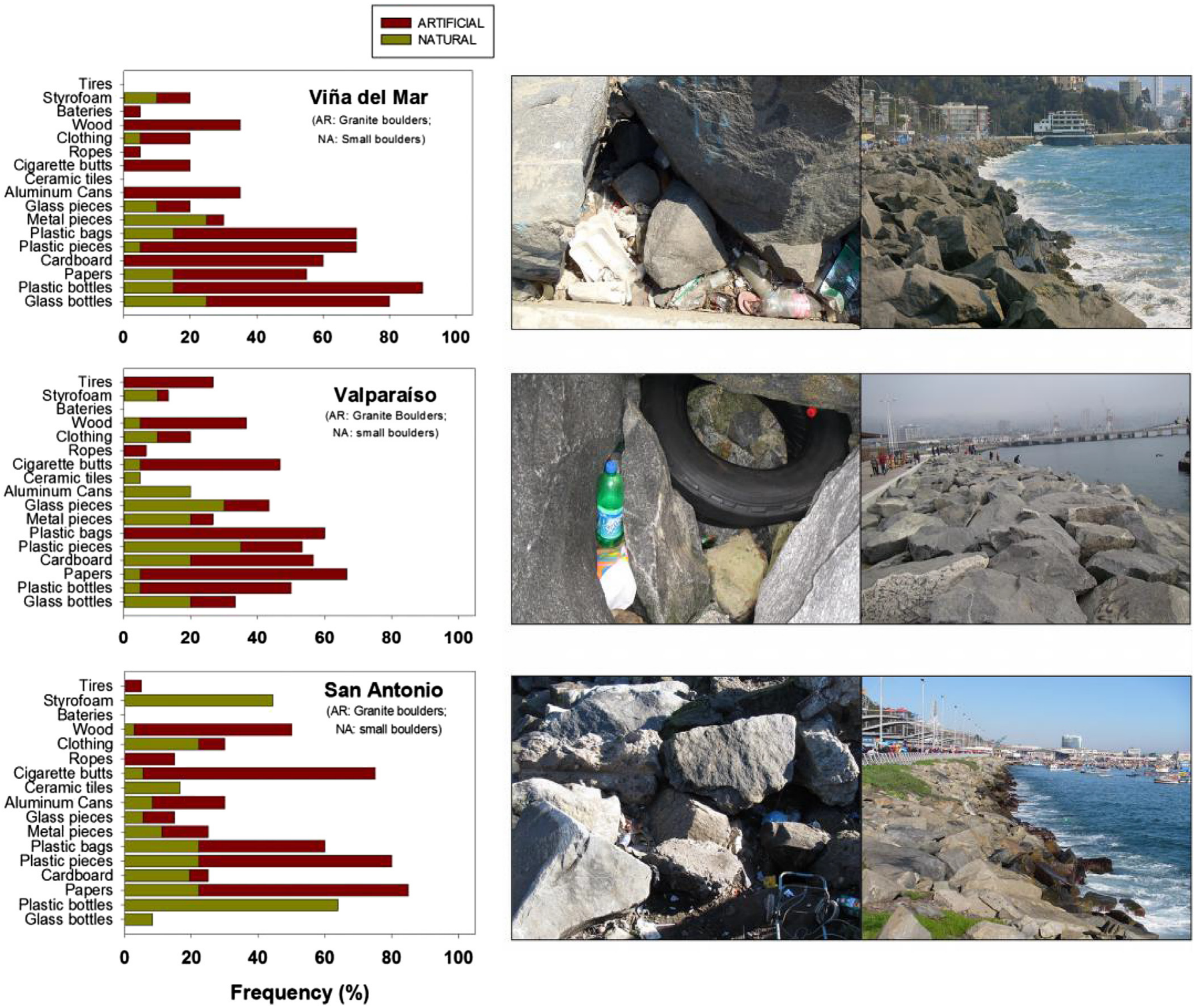

Fig. 2. (*continued*)

by rivers, which can bring AL materials from inland to coastal ecosystems (Rech et al., 2014). This suite of human activities can generate important amounts of AL, which accumulates in coastal waters and in shore habitats (Thiel et al., 2013). Breakwaters made of granite boulders (known as 'rip raps') and/or cement "tetrapods" are some of the most common building blocks for artificial reefs, having variable extension and spatial frequency across the region (see Fig. 1). Especially in northern to central Chile, increase of urban coastal infrastructures (Chilean Ministry of Public Work, MOP and see Fig. 1), and thus human pressure on the coastal ecosystems, present a challenge for conservation and future management strategies. For all these reasons we are focusing herein on litter accumulation on artificial coastal structures in northern-central Chile.

2.2. Spatiotemporal AL sampling

We estimated the density of AL material present in seven breakwaters and natural rocky shore platforms in northern (Arica, Iquique, Antofagasta, and Coquimbo) and central (Viña del Mar, Valparaíso, and San Antonio) Chile, from 18° to 33°S. We chose localities based on accessibility to breakwaters, their length (>90 m of linear length, Fig. 1) and following the presence of extensive adjacent rocky platforms to use as the natural reference habitat located within the boundaries of each city. At each locality and habitat, we deployed 35–45 contiguous quadrats of 50*50 cm surface area across two 18–23 m transects at the higher supralittoral level (3.5–4.5 MLW), perpendicular to the shoreline. Since the contact surfaces between boulders generate cavities, holes and other spatially complex elements that make it difficult to sample AL, we adopted a quadrat projection procedure: the quadrat was placed over boulders and cavities, projecting the sides of the quadrat downwards, and then we counted all AL material falling within the projected area. In each projected quadrat area, all AL material was also photographed (but not removed), and assigned to each of the following categories: plastic bottles, glass bottles, pieces of glass (>3 cm), paper (including cardboard), metal pieces, plastic bags, other plastic items (>3 cm), tires, cigarette butts, foam pieces (i.e. Expanded Polyestyrene and Extruded Polyestyrene; EPS-XPS, respectively), aluminium cans, ceramic tiles, clothes, ropes,

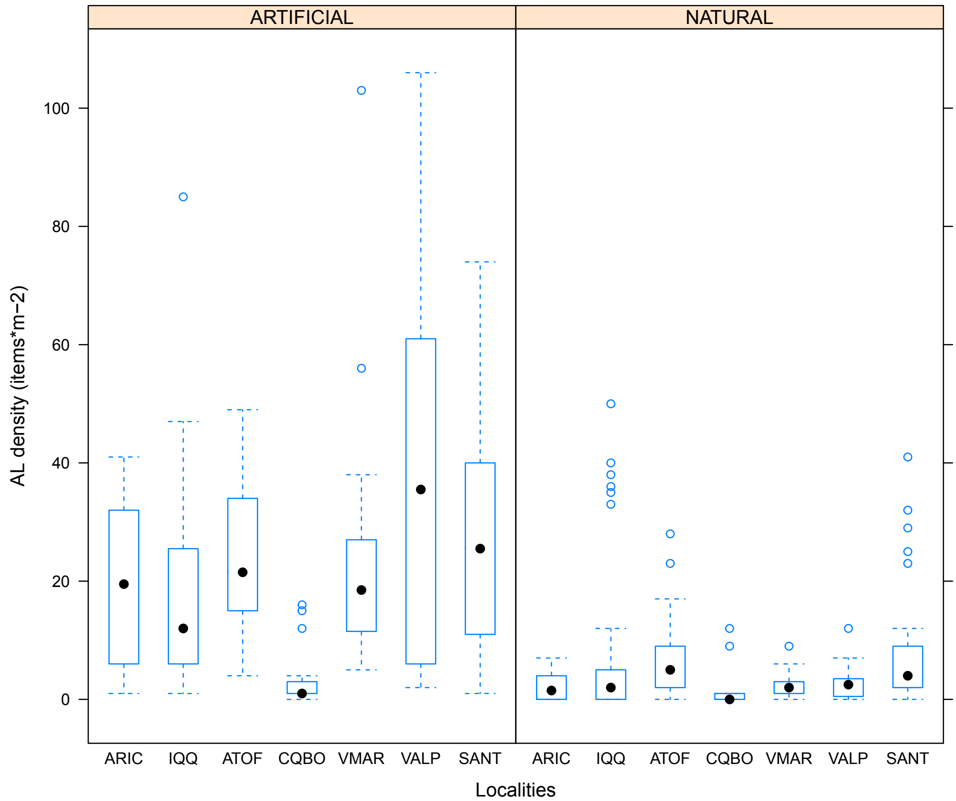

Fig. 3. Box plot of total anthropogenic litter (AL) material estimated at different localities from north (18°S) to central Chile (33°S) in both artificial breakwaters and natural rocky platforms. The black dots in each box is the median, the boxes define the hinge (25–75% quartile, and the line is 1.5 times the hinge). Points outside the interval (outliers) are represented as dots. Localities: ARIC: Arica; IQQ: Iquique; ATOF: Antofagasta; CQBO: Coquimbo; VMAR: Viña del Mar; VALP: Valparaíso; SANT: San Antonio.

Table 1
Nested analysis of variance (ANOVA) on total anthropogenic litter material (items × 0.25 m^{-2}) found in different localities and habitats (natural, artificial). Significant P-values ($\alpha = 0.01$) are presented in bold.

SV	df	SS	MS	F	P
Locality	5	7121	1424	5.793	**<0.0001**
Habitat (Locality)	6	1835	3058.4	12.441	**<0.0001**
Residual	244	59.983	245.8		

Table 2
Model selection according to Maximum Likelihood (logL) parameter estimation, Akaike and Bayesian Information Criteria (AIC and BIC, respectively) for the occurrence of total AL material. The model with the lowest information criteria is in bold. Population density (density) nested within latitude (Lat) of each city, and breakwater length (Length) were included in the full model as explanatory variables.

logL	AIC	BIC	Model
−133.611	257.290	267.430	Lat/density
−140.245	284.489	291.249	Length
−119.548	**247.096**	**260.615**	Lat/density + Length

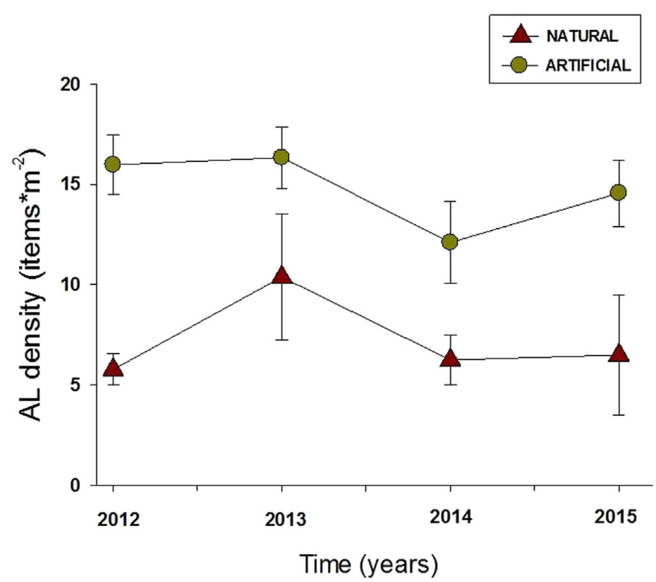

Fig. 4. Annual average density (±SE) of total anthropogenic litter (AL) material found in both artificial granite breakwater and adjacent natural rocky intertidal habitats estimated in the locality of Iquique (20°12′S).

batteries, and wood (see Fig. S1 in Supplementary Information). Local human population densities at each study locality were obtained from national census data (INE, 2012).

Each locality and habitat was sampled two times. Field campaigns were conducted from February 2013 to July 2015. In order to examine if spatial patterns of anthropogenic litter had a temporal pattern, we sampled one locality in northern Chile (Iquique) three times per year, from January 2012 to July 2015, on a total of 12 occasions.

2.3. Structural complexity

At six study sites, we estimated the structural complexity of each intertidal breakwater and natural adjacent platforms in terms of the distribution of spatial elements like the shape of granite

Table 3
Summarizing ANOVA table of the linear mixed-effect model used to analyze differences in density of anthropogenic litter material sampled at different years at the locality of Iquique. The model considered a random intercept for year (Time), and an autoregressive model of order 1(AR1) for random effects (between-Time differences). Significant P-values ($\alpha = 0.01$) are presented in bold. Parameter estimations for random and fixed effects are presented in Table S3 in the Supporting information.

SV	df	F	P
Intercept	1	150.17	**<0.0001**
Habitat	1	30.65	**<0.0001**
Time	3	0.012	0.9226
Habitat × Time	3	0.0022	0.9634

complexity of the substratum (hereafter Complexity Index; Beck, 1998). As the AL sampling consisted of contiguous quadrats (see above) we followed the same protocol for the chain-tape sampling. We deployed a 1-m long chain and tape (linear distance) multiple times along a contiguous path, i.e. the beginning of the chain (and tape) was placed on the end point of the previous chain placement (15–20 times for each transect), on the breakwater and natural adjacent platforms along the same transects (totalling 30 to 40 samples for each habitat) where AL sampling was conducted (see above). In this way, we obtained 30 to 40 measures for the complexity index for each habitat. At one of the seven study lo-

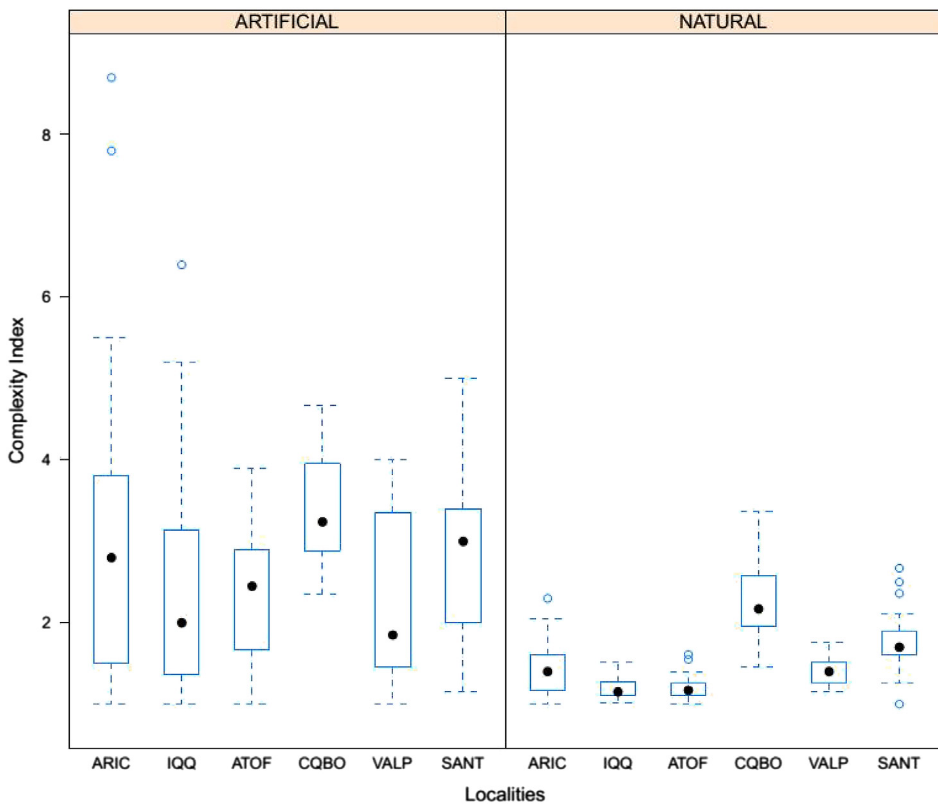

Fig. 5. Box plot of the spatial complexity index, estimated at different localities from north (18°S) to central (33°S) Chile in both artificial breakwaters and natural rocky intertidal habitats. The black dots in each box is the median, box and whiskers, and localities as in Fig. 2. Data for one locality (Viña del Mar; VMAR) are not presented here because the low sample size obtained for the artificial reef (see text for details).

Table 4
Nested analysis of variance (ANOVA) on complexity index (a measure of structural complexity) estimated at different localities and habitats (natural, artificial). Significant P-values ($\alpha = 0.01$) are presented in bold.

SV	df	SS	MS	F	P
Locality	5	28.0	5.60	13.66	**<0.001**
Habitat (Locality)	6	52.28	8.71	21.25	**<0.001**
Residual	244	100.05	0.410		

boulders, presence of crevices, rock pools, depressions, etc. (McCoy and Bell, 1991), using the chain link method (Beck, 1998). To this end, we deployed a heavy iron chain and a tape multiple times across each transect where the quadrat sampling was conducted (see above). The chain links (length = 2.5 cm) closely followed the surface and the tape only touched the highest parts of the rocks. Thus the ratio between the linear distances (tape) vs. the apparent distance (chain) can be considered as a measure of the structural

calities this sampling could only be partially completed (Viña del Mar), given that the strong and persistent wave surge and slope of the breakwaters at the sampling time (August–December 2015) restricted the access to this site. Thus, we only include partial data sets (10 measures of the Complexity Index) for this locality for further spatial analyses (see below).

Given inherent differences between intertidal breakwaters (e.g. made of granite boulders versus cement tetrapods see Fig. 2), we first conducted a pilot study using three different chain link size; 2.5 cm, 4 cm, and 6 cm per link. We determined that a chain link of 2.5 cm was sufficient to characterize the spatial distribution of both granite boulders and tetrapods at the study sites considered, and consequently all results reported are based on this link size.

2.4. Statistical analyses

Spatial dependence, i.e. spatial autocorrelation, of AL densities

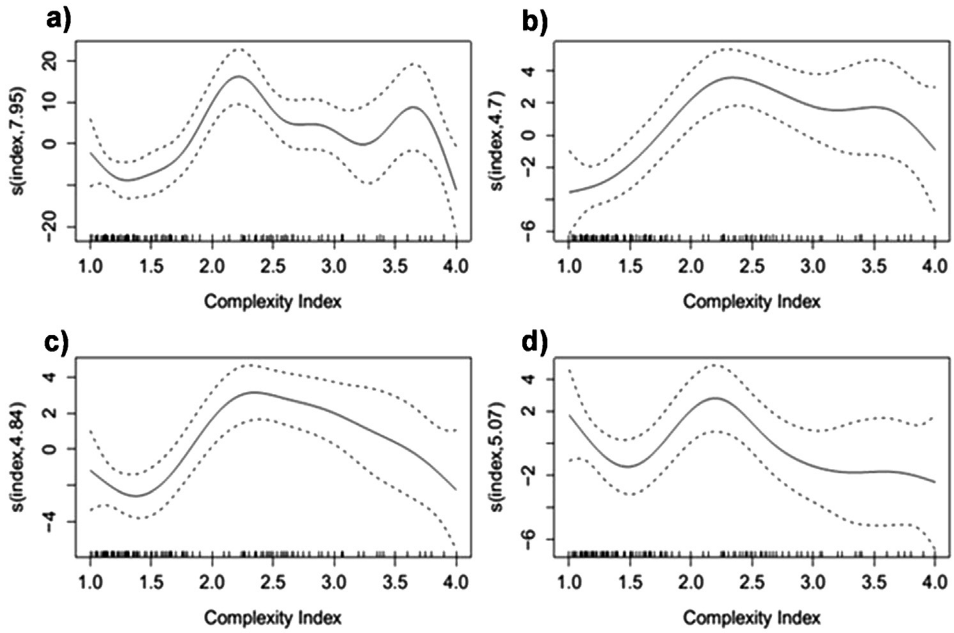

Fig. 6. Estimated smoothing curves for a) Total AL density, b) Plastics (bags, bottles, and small pieces), c) papers and d) glass material density. The solid line is the smoother and the dotted lines are 95% point-wise confidence bands.

Table 5
Summary table for GAM model (Gaussian distribution with identity link function), for different anthropogenic litter (AL) density categories (pooled items) as response variables, and complexity index as the explanatory variable. A cubic smoothing term was considered in the model. Adjusted r^2, effective degree of freedom (edf), P-value and deviance explained by the smooth function are shown, as well as the generalized cross validation (GCV) score for the model. Significant P-values ($\alpha = 0.01$) are presented in bold.

Parameters	Total AL	Plastic	Papers	Glasses
Adjusted r^2	0.20	0.15	0.12	0.036
Deviance Explained (%)	35.0	16.3	14.5	6.5
GCV score	290.98	50.92	35.70	57.2
edf	7.945	4.703	4.839	5.067
P-value	$\mathbf{9.63 \times 10^{-8}}$	$\mathbf{6.04 \times 10^{-6}}$	$\mathbf{6.68 \times 10^{-6}}$	0.0784

can be expected to arise from our sampling procedure, thus precluding most parametric analyses (Fortin and Dale, 2005). To this end, we first explored if density of the most important AL categories (i.e. plastic bags, plastic and glass bottles, plastic pieces, papers, cardboards) were spatially autocorrelated in the different habitats and localities. We estimated Moran's *I* spatial autocorrelation (Fortin and Dale, 2005) for each AL category and the structural complexity index across transects separately for artificial breakwaters and natural habitats. Statistical significance of autocorrelation coefficients was determined by means of permutations (Manly, 2006), where transect data were randomly shuffled and the autocorrelation statistic was recalculated 1000 times for each spatial lag. We pooled the density of each AL category and the complexity index found at each distance class across transects for the different localities (see summary Table S1 in Supplementary Information). We found no evidence of spatial autocorrelation in our response and/or explanatory variables across localities (see Table S1).

In order to test whether human population density affects the occurrence (i.e. presence/absence across sampling units) of AL (including all litter categories) recorded on the different breakwaters, we conducted a multiple logistic regression analysis, with population density at each sampled locality and breakwater linear length as explanatory variables. Local human population density was considered random and nested in each of the seven localities (cities) considered for analysis. This analysis allows to explore the chance that a quadrat has AL or not. Before the analyses, we computed for each explanatory variable its variance inflation factor (VIF) in order to detect collinearity (Zuur et al., 2009). The term with highest VIF was sequentially removed until all remaining terms showed VIF <2. Akaike and Bayesian information criteria (AIC and BIC, respectively) were used to find the best fit to the probability of occurrence (i.e. presence/absence across sampling units) of AL materials.

Differences in the density of AL observed between the two different habitat types were analysed using a two-way nested ANOVA, considering localities as random factor and natural and artificial reef habitats as fixed factors nested within localities. Between-habitat differences for the long-term AL survey conducted at one study locality (Iquique) were analysed using a linear mixed-effect model with habitat as fixed factor and time as a random factor, as before. Thus, we considered a random intercept model (obtained with a maximum likelihood estimate), considering an autoregressive model of order 1 (AR1) as the residual autocorrelation structure of AL through time (i.e. three sampling dates within four years) (Zuur et al., 2009). Homogeneity of variances was graphically explored by means of residuals-vs.-fits and normal Q-Q plots. Structural complexity estimated at different sites for natural and artificial habitats was also analysed with a nested ANOVA as before. All data met the assumptions of normality and homoscedasticity.

We examined the relationship between the abundance of individual AL categories and the index of structural complexity estimated at each habitat and locality. In order to standardize estimates of the complexity index obtained, we used 30 measures for each habitat; i.e. 60 per locality. Exclusively for the locality of Viña del Mar, we only used 10 estimates obtained on the breakwater (see above) and 30 on the natural habitat. Thus, we considered a total of 400 estimates of the complexity index, and corresponding AL material, for all seven sampled localities for analyses. An *a priori* check of the distribution of density of AL and complexity index across habitats and localities suggested a more complex, non-linear pattern. Thus, we used a generalized additive model (GAM) considering a Gaussian distribution of AL material with an identity link function. We used the complexity index values estimated in each habitat (at seven localities) as the explanatory variable, considering a cubic regression spline as the smoothing function (Zuur et al., 2009). All analyses were conducted in the R environment version 3.1.3 (R Development Core TeamR., 2011).

3. Results

3.1. Frequency of breakwaters across Chile

More than 299.3 km of the Chilean coastline correspond to breakwaters or rip raps made of granite boulders and tetrapods which were built to bolster pedestrian promenades, harbours and artificial beaches (Fig. 1). The average linear length of individual breakwaters was 431.5 (±142 SD) meters, but their lengths are highly variable across the Chilean coast from 18°S to 41°S (Fig. 1). The cities with the longest artificial coastlines were Puerto Montt (approx. 4.32 km), Talcahuano (2.99 km), and Arica (2.84 km).

3.2. AL distribution in natural and artificial reefs

At all study sites the density of AL was significantly higher on breakwaters when compared with adjacent natural habitats (Fig. 2 and 3, Table 1). The AL items that most contributed to these differences (70% of between-habitat differences) were plastic bags, paper and cardboard, plastic pieces, glass and plastic bottles (Fig. 2). All these items were abundant on the artificial reefs compared with natural rocky platforms for most localities, except at the locality of San Antonio where glass and plastic bottles were higher in natural habitats (see Figs. 2 and 3).

In addition, we found that breakwater extension (linear length) and local population density contribute significantly to the probability of AL occurrence (i.e. increase in the probability of AL presence across sampling units, see Table S2 in Supplementary Information). Akaike and Bayesian information criteria were smallest for the model that included breakwater length, and human population density nested within latitude as explanatory variables (Table 2). The odds ratio for breakwater length was estimated as 0.997 with 95% CI ranging from 0.995 to 1.00 (Table S2), indicating that for one unit (i.e. metres) of increase in breakwater length, a quadrat (sampling unit) had ~1.0 chance of having AL compared to not having AL items. This corresponds to an increase in the odds of having AL items, for each metre of each breakwater, of ~10%. A relatively similar pattern was found for population density nested within latitudes, with each increase in hab./m^2 increasing the odds of AL items by ~10% (Table S2).

Long-term sampling at one of the studied localities (Iquique, 20°11′S) from January 2012 to August 2015 showed that AL density on the breakwater (granite boulders) was significantly higher than in natural habitats and this pattern was persistent over time (Fig. 4, Table 3 and Table S3 in Supplementary Information). AL density on the Iquique breakwater had an average (four-year data) of 15.1 (±0.96) items m^{-2}, while the adjacent natural habitat (rocky platform) had 7.4 (±2.19) items m^{-2}. Thus, the density of AL was consistently higher on the artificial reef than in the natural habitat throughout the four years of study at this site (Fig. 4, Table 3).

3.3. Structural complexity and AL distribution

Structural complexity significantly differed between breakwaters and natural habitats at the different localities considered (Fig. 5, Table 4). At all sites, breakwaters had higher complexity index values compared with natural habitats, in which the values were close to 1 (i.e. a more uniform structural complexity at the scale considered, Fig. 5). Coastal breakwaters sampled in Arica (tetrapods) and Antofagasta (granite boulders) showed the highest median values of structural complexity (2.8 and 2.5, respectively), with the former having a maximum index value of 8.7 (see outliers in Fig. 5).

Our generalized additive model (GAM) indicated that total AL density, estimated across seven sampling localities and habitats, had a significant non-linear relationship with the structural complexity (Fig. 6a, Table 5). At intermediate structural complexity, with index values ranging from 1.9 to 2.5, typical for most urban breakwaters (see Fig. 5), total AL density was highest (Fig. 6a). Similarly, total plastic material (i.e. bags, bottles, small pieces) and paper (including cardboards) showed a relatively similar pattern, increasing at intermediate structural complexity (Fig. 6b and c). Despite the fact that the model revealed significant relationships for these AL categories (Table 5), deviance explained by the model was lower for plastics and paper (16.3, 14.5%, respectively) than for total AL (i.e. 35%; Table 5). This result arose from the poor fit of the model at high structural complexity (typical for tetrapods) for these AL categories, which is confirmed by the wide confidence intervals observed at the complexity levels (see Fig. 6b and c). Anyways, the smooth function of the model was significant for these AL categories (Table 5). No significant estimates were obtained for glass (bottles and small pieces), and the model only explained 6.5% of total deviance in glass density (Table 5).

4. Discussion

Our study showed that artificial reefs, like urban granite breakwaters, harboured significantly more Anthropogenic litter (AL) than natural, adjacent, rocky shore platforms. More importantly, we found that the pattern of intermediate to high structural complexity on artificial reefs was a significant predictor of an increase in AL density. Complementary, we also found evidence that human population size and breakwater extension positively affected the probability of AL occurrence. Here, we discuss our results in relation to ecological and human population consequences of coastal engineering infrastructure and the management opportunities provided by structural characteristics of the artificial and natural habitats in coastal ecosystems.

4.1. Origin, composition and persistence of AL on breakwaters

The AL categories recorded at the different localities along the coast of northern-central Chile were very similar between breakwaters and rocky shore habitats. The different AL categories observed on the artificial reefs and in natural rocky habitats were also relatively similar to those observed in rocky shore habitats by Thiel et al. (2013) in areas near fishing ports. Studies conducted on different sandy beaches across the Chilean coast by Bravo et al. (2009) found substantially higher proportions of metal, and in particular of glass. Instead, we found at most sites higher proportions of lighter material, such as plastic (i.e. small pieces, bags, and bottles), paper and cardboard, with a low proportion of metal. Exceptionally, some localities showed a high proportion of wood and clothes (Arica), ropes (Coquimbo), and tires (Valparaiso, San Antonio), which can be related to specific activities conducted at each breakwater (see below).

Plastics and papers have been reported to be common in rocky shore habitats and coastal waters of northern-central Chile (Thiel et al., 2013). Buoyant elements like plastic material are more susceptible to become trapped in breakwaters when floating in coastal waters, thereby contributing to AL accumulation in coastal habitats (Thiel et al., 2013). Rivers are one of the main source transporting litter from inland sources to the coast (Rech et al., 2014) thereby contributing to higher densities of AL in zones adjacent to the river mouth (De Araújo and da Costa, 2007). For example, the natural habitats in the southernmost locality sampled (San Antonio), which is located few km north of a large river mouth (Maipo), had higher amounts of plastic bottles and Styrofoam (both Expanded Polystyrene (EPS) and Extruded Polystyrene (XPS)) compared with the breakwater, also located near the river mouth, at this same locality. This can be related to the input of AL from the river, which imports floating AL to coastal ecosystems around the river mouth (Rech et al., 2014).

Our results are concordant with studies from other coasts reporting a direct relation between AL abundance (e.g. microplastic) and human population (Browne et al., 2011). In this context, given the origin of the AL elements found on breakwaters, accumulation and/or retention can be mostly related to local human activities on, or near, the artificial reef. Throughout the study, we observed that people frequently developed a range of activities on the different breakwaters (i.e. camping, fishing, eating, leisure, etc.), which could be related to the specific AL material found. Most urban breakwaters sampled were located near or built for pedestrian promenades, residential buildings and artificial beaches. Thus, these artificial reefs have also a value for recreational activities (Hill, 2015), but are susceptible to pollution derived from human activities. In this context, we found evidence that breakwater length and local population density can increase total AL occurrence. This may be related to more intense and diverse human activities taking place on longer breakwaters which is an aspect that needs future research.

The importance of urban breakwaters as coastal recreational area and their structural complexity seems to greatly account for higher occurrences, as shown by relationships with human population size. Also, densities of AL found on breakwaters compared with natural rocky (Thiel et al., 2013; this study) and sandy beach habitats (Bravo et al., 2009; Thiel et al., 2011) reported for the South East Pacific coasts. In this way, incorporating a human dimension in the design of shoreline infrastructure may also contribute to enhance urban coastal habitats by taking into account the multiple services that society derives from them (Abelson et al., 2015; Hill, 2015).

Spatial distribution of AL was captured well by our contiguous sampling protocol on the complex boulder structure, and also on the small-scale topographic complexity of natural rocky habitats (crevices, pools, etc.). This allowed us to determine that among-boulders spaces and cavities that are generated when granite breakwaters are built are the main factor influencing AL accumulation compared with natural habitats. Thus, our results underline the need to investigate the dynamics and accumulation rates of AL on artificial and natural rocky habitats by considering elements of the spatial structure of natural and artificial habitats.

4.2. Anthropogenic litter and potential for ecological effects on artificial reefs

Given that no litter removal strategies are currently applied to the coastal infrastructures surveyed, it is expected that artificial breakwaters (i.e. granite boulders) enhance residence time of most AL material accumulating on them. Especially, we found that different litter elements were retained between boulders and/or inside the cavities (about 50–100 cm wide) between the contact surfaces of boulders on artificial breakwaters. Neither strong wave action during incoming high tide nor wind action seemed to remove materials from this particular habitat over the study period (MAA, personal observations). However, high waves actions could remove AL material from breakwaters to other habitats thus reducing accumulation and some AL categories could be more susceptible to wave removal than other (e.g. plastic bottles compared with paper or metal) an issue which deserve further research. This seems to explain the direct relation between total anthropogenic litter accumulation and intermediate structural complexity we found for the artificial habitat. Density of the different categories of AL were related to intermediate structural complexity levels, and were relatively less dense at higher complexity levels characteristic of tetrapods which had poor resolution in our model. Given that among-tetrapods contact (as well as some parts of the granite breakwaters) conforms ample spaces (about 1 m wide), low and/or variable retention of AL material is expected at higher scales of structural complexity. Thus, high spatial variability in some AL categories is expected at these levels of structural complexity characteristic of tetrapods. Anyways, we found the model captured well the non-linear relationship between structural complexity and most AL material in the study habitats.

Plastic pieces (3–6 cm), plastic bottles and bags were common items on breakwaters at all study sites, with an increase in density at intermediate structural complexity levels. Therefore, accumulation of these elements in the breakwaters may have negative impacts on the benthic community (Green et al., 2015).

In marine habitats, detritivores and filter feeders can be critically affected by consumption of microplastic material (Browne et al., 2015; Thompson and Moore, 2009; Wright et al., 2013). For example, the persistence of some filter feeder species like mussels, which are absent on artificial reefs (Aguilera et al., 2014; Firth et al., 2014), may be affected by AL presence and its degradation. In contrast, other species can benefit by AL accumulation on breakwaters which seems to provide them with shelter and/or food (see Katsanevakis et al., 2007). For example, we observed higher abundances of crabs (*Leptograpsus* spp.) and rats (e.g. *Rattus norvegicus*) on breakwaters compared with adjacent natural rocky platforms (Crabs: 1.8 individuals m^{-2} on the breakwater versus 0.07 ind. m^{-2} in natural habitats; Rats: 0.3 individuals m^{-2} versus none in natural habitats; MAA, unpublished data). These species use the small scale complexity generated by AL elements for shelter, and consume organic AL material (MAA, personal observations). However, while these observations are highly suggestive, it is not entirely clear if the combined or isolated effect of these resources contribute to the high abundances of crabs and rats in the artificial habitats. In addition, breakwaters can trap floating AL elements colonized with non-indigenous species, thereby possibly facilitating their spread (Kiessling et al., 2015). Field experiments are needed to determine how AL affects species richness, life-history stages and successional phases of the hard-bottom community, thereby potentially causing a decline in the functioning of the ecosystem, and opening a “window of opportunities” for species invasions (see Bulleri and Airoldi, 2005).

4.3. Ameliorating the impacts of coastal cities, engineering of the spatial structure of artificial reefs

Although social and ecological costs of degraded habitats are recognized (Scyphers et al., 2014), the armouring of shorelines has continued apace (Gittman et al., 2015; Perkins et al., 2015). The extent of artificial breakwaters in Chile (4.5% of the total shoreline length) is still low compared with other coasts (e.g. Florida, USA - Gittman et al., 2015; Singapore - Lai et al., 2015). However, coastal infrastructure in Chile is expanding (MOP, and see Fig. 1) constituting a challenge for future improvement of altered coastal ecosystems. The recovery and improvement of altered engineered coastlines in the future (Abelson et al., 2015; Scyphers et al., 2014), and also their re-design by means of ecologically-based information (i.e. ecological engineering; Browne and Chapman, 2011; Chapman and Underwood, 2011; Chapman and Blockley, 2009) requires immediate action.

Information of the qualitative characteristics of AL (i.e. if this is from medical, industrial origin, or if have toxic elements) can provide a basic guideline for cleaning strategies and health security (Thompson, 2015). Notwithstanding, initiatives to reduce AL at the source may be more effective. Complimentary with this, management plans could also incorporate more ecologically sustainable infrastructure designs that enhance biodiversity (e.g. Firth et al., 2014; Martins et al., 2015; Perkins et al., 2015), and reduce AL accumulation. Our results indicate that modifying structural complexity, e.g. reducing wide cavities among boulders, would reduce accumulation of AL in the zones that are most influenced by direct anthropogenic activities. Since AL material also enter breakwaters as floating debris (Hinojosa and Thiel, 2009), such mitigation initiatives would be partially effective. Therefore, there is a challenge for future structural designs of artificial reefs (e.g. Hill, 2015; Perkins et al., 2015), which should also take into account the mitigation of AL accumulation on them.

5. Conclusions

Current policies to control and to prevent costs and impacts of coastal modification by urbanization are defined by social and ecological information (Scyphers et al., 2014). Knowledge of emergent impacts of urbanization, as presented herein, is essential for designing effective ecosystem restoration plans (Abelson et al., 2015). We showed that artificial reefs can have the potential to degrade local habitats by enhancing retention and accumulation of human-derived litter material. Specifically, we found that the structural complexity at scales of dozens of centimetres to few meters (e.g. made by spaces between artificial boulders) contributes to the high density of AL on urban breakwaters, which is further enhanced by both adjacent urban activities and entrapment of floating debris. Therefore, the direct relationship between habitat modification and pollution seems to be an emergent phenomenon related to urbanization of coastal ecosystems. The environmental impacts of AL material and infrastructures derived from urbanization are still emerging areas of research. Future work on this issue will contribute to the development of better methods for controlling and re-designing coastal infrastructures providing them with ample ecological value.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.envpol.2016.04.058.

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