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ECOENG-4702; No. of Pages 10
ARTICLE IN PRESS

Ecological Engineering xxx (2017) xxx–xxx

Contents lists available at ScienceDirect

Ecological Engineering

journal homepage: www.elsevier.com/locate/ecoleng

Artificial defences in coastal marine ecosystems in Chile: Opportunities for spatial planning to mitigate habitat loss and alteration of the marine community structure

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ARTICLE INFO

Article history:
Received 19 October 2016
Received in revised form 23 March 2017
Accepted 6 April 2017
Available online xxx

Keywords:
Habitat loss
Coastal ecosystem
Functional structure
Rehabilitation
Chile

ABSTRACT

Many coastal habitats are actually replaced with hard infrastructures which alter the taxonomic/functional structure of natural ecosystems worldwide. Few information about habitat loss and species composition in South American coasts are available compared with other coasts. Here, I examine the distribution and identity of coastal artificial infrastructures, especially artificial breakwaters, present along the coast of Chile and the proportion of natural habitat loss derived from their construction. Differences in species taxonomic/functional composition in artificial breakwaters and natural habitats present in northern Chile are also examined. I also propose/discuss opportunities for coastal planning based on habitat rehabilitation and ecological engineering in Chile, which could guide future marine infrastructures construction. An important proportion of natural habitat has been replaced by artificial coastal defences along the coast of Chile, accounting for about 200 km of total coastal length. Given their specific uses and functions, artificial granite breakwaters are one of the most important coastal infrastructures present in Chile (62% of the total of artificial breakwaters present). Differences in taxonomic/functional structure between artificial breakwaters and natural adjacent habitats are significant, and appear related to contrasting spatial heterogeneity. Artificial infrastructures like granite breakwaters can facilitate presence of native and non-native species, which live in the marine-terrestrial interphase (crabs, rats). The present study highlights how the recent proliferation of coastal artificial infrastructures is replacing important natural habitats in Chile, and how the taxonomic/functional structure of coastal ecosystems can be negatively impacted. Furthermore, this study showed how artificial infrastructures can have direct consequences for human-health security and specific guidelines can be conducted to buffer impacts on ecosystem structure to match social livelihood and wellness.

1. Introduction

Increasing human population growth is enhancing the demands for natural coastal habitats around the world, and many coastal habitats are replaced with hard infrastructures which alter the structure and functioning of marine ecosystems (Airoldi et al., 2005; Bulleri and Chapman, 2010; Bulleri, 2006; Connell and Glasby, 1999; Firth et al., 2016). As coastal habitats are an important interphase between land and the sea, their modification by infrastructure construction can affect the ecosystem potential for disturbance recovery given reduction of key habitats and alteration of species composition (Bulleri and Chapman, 2010; Chapman and Underwood, 2011; Firth et al., 2016; Waltham and Sheaves, 2015). For example, enhancement of exotic species and loss of native ones in artificial infrastructures have been reported in different coasts (e.g. Italy: Airoldi et al., 2015; Bulleri and Airoldi, 2005; Australia: Chapman and Underwood, 2011; Browne and Chapman, 2011; Singapore: Lai et al., 2015; UK: Firth et al., 2014), which could result in loss of important ecosystem functions and services (Dafforn et al., 2015b). However, few studies have examined the spatial processes determining variation in assemblages' functional structure in natural versus artificial habitats (see Firth et al., 2014) which account for alteration of ecosystem functioning. In this context, there exists considerable evidence showing regional reduction of particular biogenic habitats (e.g. mussel beds, kelps, coral reefs) across the world related to the presence of artificial infrastructures (reviewed by Firth et al., 2016). This loss of natural coastal defences (e.g. Coral reefs, Ferrario et al., 2014) can have critical consequences

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http://dx.doi.org/10.1016/j.ecoleng.2017.04.021
0925-8574/

on ecosystem services like wave attenuation or reduction of flooding risk (see Bouma et al., 2014 for review). Thus, urgent actions are required globally before local ecosystems shift to an unproductive state with low capacity to recovery from anthropogenic and natural disturbances (Dafforn et al., 2015a).

In general, most information dealing with reduction of habitat or loss of natural coastal defences by infrastructure construction come from Japan, Europe (especially UK and Italy), Australia and North America (e.g. Baine, 2001; Evans et al., 2016; Firth et al., 2016). Little information about 'ocean sprawl' rates and consequent habitat loss in South American coasts is available (Aguilera et al., 2014; Defeo et al., 2009; Jaramillo et al., 2002). Along the Chilean coast (18°S–56°S), presence and extension of coastal artificial infrastructures has been increasing in coastal cities in parallel to the higher frequency of storms and tsunamis over the last 10 years (DIRECTEMAR, 2016). Thus, higher governmental investment for their construction is actually considered (MOP-DOP, 2016). As in other emergent economies, plans to expand coastal development (mining, aquaculture) across Chile means that the risk of damage from anthropogenic stresses is imminent (see Gittman et al., 2015; He et al., 2014; Waltham and Sheaves, 2015). Given that most infrastructures have been recently built in Chile (from 2000 to date), little attention has been paid to their impacts on coastal ecosystems. Thus, no management plans or policies are available to guide habitat rehabilitation or infrastructure re-design. The geographic extension and ecosystem diversity present along the Chilean coast constitutes a challenge to develop integrative management plans and appropriated governmental policies for future 'ocean sprawl'.

In this study, I examine the distribution of coastal artificial infrastructures present along the coast of Chile from 18°S to 41°S, and the proportion of natural habitat loss derived from their construction. As study case, the main differences in consumer species abundance, and consumer and sessile assemblage functional composition in artificial breakwaters and natural habitats present in northern Chile (20°S) are also evaluated. This information could contribute to develop future coastal infrastructure planning based on specific characteristics of natural habitats, and their community structure. Thus, opportunities for coastal planning and ecological engineering are also discussed to guide future marine infrastructures construction in Chile.

2. Material and methods

2.1. Distribution of artificial breakwaters across the Chilean coast, and replacement of natural habitats

In order to estimate the length of the coastline covered with different artificial infrastructures, and to quantify the different types of artificial breakwaters present across the coast of Chile, I obtained information from different localities (cities) from Arica to Puerto Montt (i.e. 18°S–41°S) documenting location and length of each breakwater and the previous natural habitats present prior to construction (i.e. replaced habitat). In this case only infrastructures built until October 2016 were included. Information was obtained from different sources: (1) from Chilean Ministry of Public Works (MOP-DOP, 2016), (2) personal visit and length estimation of each breakwater, complementing with (3) estimations of the linear length of each breakwater using tools options provided by the program Google Earth v7. Geographic location for each site was determined using GPS option in Google Earth (see also Aguilera et al., 2016; and Waltham and Sheaves, 2015 for effectivity of this method).

2.2. Variation in species composition in natural versus artificial breakwaters in intertidal and supralittoral levels

2.2.1. Sampling for slow-moving invertebrate species and sessile organism

The density of invertebrate species like limpets, chitons, echinoderms, and the percentage cover of sessile organisms (algae, sessile invertebrates) was estimated through a quadrat-sampling protocol in an urban granite breakwater, and in natural adjacent habitats present in northern Chile (Iquique, 20°S). The study artificial breakwater was built in 2005 to stabilize a public walk, and was made of granite boulders about of 1.3 m^2 (±0.42) in size. In both habitats, I consider long-term surveys from 2009 to 2013 (see Aguilera et al., 2014), and from 2014 to 2015. Sampling was conducted seasonally (each 4 month). In these surveys, sampling protocols were as follow; along the breakwater I deployed 25–35 quadrats of 30 * 30 cm surface area that were separated from each other by 30 cm, at both the mid and high intertidal zones corresponding to the supralittoral fringe (4.5–6.0 m above chart datum). On two adjacent natural rocky platforms (15–20 m long), on either side of the breakwater and distant from each other by approximately 150 m, we deployed 25–30 quadrats at the mid and high intertidal level as before. We took photographs using a digital camera positioned directly above each quadrat (Foster et al., 1991). Each photograph was cropped to include only the quadrat and analysed using the program image J (http://imagej.-nih.gov/ij/download.html). Percentage cover for sessile organisms was quantified by projecting 25 dots randomly onto each photo, and assigning a value of 4% to each organism that occurred in each dot. Photographs were analysed digitally using a high-pass filter method for abundance estimation of benthic organisms to improve sampling (Pech et al., 2004). This method eliminates distortion caused by panoramic effects like shadows and changes in rock surface coloration, thus increasing geometric details of each photograph.

2.2.2. Sampling protocol for fast-moving species

In order to estimate the density of highly mobile species like crabs (*Grapsus* spp.), as well as other animals like rats and cats observed in the artificial breakwater (see Results) and the natural habitat, I use a non-invasive sampling protocol (i.e. trap-sampling was forbidden by local authorities). Commonly crabs inhabiting higher intertidal and supralittoral levels are sensitive to humans when following their foraging activities (during low tide), thus crabs take-flight quickly under human presence (Abele et al., 1986). Along the artificial breakwater and in two adjacent natural rocky platforms, we deploy two perpendicular 35 m long measurement tapes that were separated from each other by two meters. Each two meters along the transect, and distanced away from it from about three meters, we took photographs of the granite boulders and rocky substratum present in the middle of both perpendicular measure tapes. To reduce the angle of each photographs, a person (1.74 m tall) mounted in a 1 m-tall object to take photographs. All photographs were taken 15 min after the deployment of the measurement tape, allowing animals (especially crabs) to return to normal foraging activities i.e. when any person was present in the platform. Thus, in each habitat type we obtained 10 photographs 4*4 m surface area in which we estimated the number of individual of each mobile species present. To complement this sampling in the artificial breakwater, we took photographs among-boulder spaces in each area to detect the proportion of individuals sheltering on them. Samplings were conducted each 4 months from January 2013 to April 2016, from mid-day to afternoon according to spring and neap tides.