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Effects of artificial defences and flooding on coastal habitats and assemblages

Candidato: **Giulio Franzitta**

Settore scientifico disciplinare BIO/07
Settore concorsuale 05/C1

Coordinatore:
Prof. **Enrico Dinelli**

Tutore:
Prof. **Marco Abbiati**

Relatore:
Prof.ssa **Laura Airoldi**

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Your woraciousness, fellow-critters.
I don't blame ye so much for;
dat is natur, and can't be helped;
but to gobern dat wicked natur, dat is de pint.
You is sharks, sartin;
but if you gobern de shark in you, why den you be angel;
for all angel is not'ing more dan de shark well goberned.

Herman Melville, Moby Dick, 1851.

CHAPTER 1

General introduction

Proliferation of coastal defence structures

Historically, coastal zones have played a key role for human societies and economies. This territory has been always used by human populations for settlement, trades and the utilization of natural resources.

Estuaries, lagoons, rocky shores and beaches have experienced a dramatic transformation over the past decades, which is still ongoing, particularly along the European coasts. The origin of those transformations are both natural and anthropogenic, and it is largely recognized that the global climate change is accelerating those processes.

The anthropogenic alteration of the coastal environment is caused by multiple factors, among which the urbanization. In fact, nowadays many of the larger European cities (e.g. Barcelona, Athens, Istanbul, Tripoli) are built along the coast, or (as in the case of London, Hamburg, St. Petersburg, and Thessaloniki) on estuaries and lagoons. There are 15 port and industrial coastal cities with more than 1 million people within Europe (Nicholls et al., 2007). Many industrial plants are built near the shore, as well as infrastructures and facilities for tourism, fishing, fish farming and agriculture.

The Mediterranean is a good example of a coastal region where urban development is already significant and continues to grow. The population density along Mediterranean coast was estimated to be about 6000 people for km of coastline (UNEP/MAP/PAP, 2001). In twenty years (1980-2000) the human population living at the coast grew by the 46 % exceeding 123 million in the 2000, and it is projected to nearly double between 2000 and 2025 (UNEP/MAP/PAP, 2001). The Mediterranean coastline is also intensively utilized for coastal tourism, accounting for 30 % of the global tourism, that are project to reach 235–350 million tourists by the 2025 (EEA, 1999). This means that significant amounts of recreational infrastructure already exist, or will be built, immediately adjacent to the coast. Indeed, two thirds of the Mediterranean coastline is already urbanized (UNEP/MAP/PAP, 2001) and more than 50 % of the Mediterranean coasts are dominated by concrete with

more than 1500 km of artificial coasts (EEA, 1999). Overall coastal zone urbanization is projected to increase by 10–20% for most Mediterranean countries (EEA, 2006).

This situation is common to many other coastal areas in the world (Dugan et al., 2011). In the USA armouring covers more than 50 % of the coastline in a number of estuaries and bays (Living Shoreline Summit Steering Committee, 2006), and about 21 % of the 759 km coastlines of Florida, 12 % of the 1763 km coastline of California and 17 % of the coastline of New Jersey have been altered by either armouring or addition of bulkheads, revetments or other coastal structures (Florida DEP, 1990; Griggs, 1998; Lathrop & Love, 2007). Similar examples occur in the Western Pacific, where 27 % of the coastline in Japan (Koike, 1996) and more than 50 % of the shores of Sydney Harbour (Chapman & Bulleri, 2003) have been altered by either coastal infrastructure or armouring. It is expected that armoring will further increase as a result of burgeoning coastal populations, expansion of coastal cities, and greater threats from climate change, storm surges and sea level rise (Inger et al., 2009; Shepard et al., 2011).

Coastal defence structures as novel substrata for biota and implications at local and regional scales

Amongst the most abundant artificial structure along the coastlines all over the world are hard coastal-defences. Since coastal areas are important for industries, settlements and societies, there is a pressing demand to protect the coast from erosion, storm and flooding. Consequently, human-made defence structures of different typologies (e.g. breakwaters, groynes, seawalls, dykes or other rock armoured structures) have proliferated, becoming a common feature of the coastal landscapes in intertidal and shallow subtidal environments (National Institute of Coastal and Marine Management of the Netherlands, 2004). The primary purposes of defence structures are to prevent or reduce erosion and flooding of high value coastlines, to stabilize and retain beaches and reclaimed land, and to increase the amenity value of the coast (e.g. beach use, surfing). In Europe more than 7600 km of coast are protected or artificially stabilized (EC, 2004). In Italy, along the north-west Adriatic coast, artificial defence structures cover over half of the shoreline resulting in dramatic changes to coastal landscapes and environments (Airoldi & Beck, 2007).

Surprisingly relatively little attention has been given to the ecological implications of hard coastal-defence structures (see Bulleri & Chapman, 2010 for review). Bulleri & Chapman (2010) suggested that insufficient scientific information led to overlook urbanization and

artificial structures as major causes of anthropogenic changes in coastal ecosystems, even in the highly cited review of human impacts on marine ecosystems by Halpern et al. (2008). However, according to the current knowledge, the ecological impacts of artificial structures on coastal habitats are varied and severe, e.g.: the introduction of novel habitat for sessile and mobile species, with potential changes in the pattern of their distribution at local to regional levels (Glasby & Connell, 1999; Connell, 2000; Bacchiocchi & Airoldi, 2003; Moschella et al., 2005; Bulleri & Chapman, 2010); effects on the adjacent native sedimentary habitats, related to changes in water flow, illumination, rates of sedimentation detrital patways, with consequent negative impacts on the benthic infauna (Glasby, 1999; Martin et al., 2005; Bertasi et al., 2007); the local loss of species of particular functional groups, e.g. large grazers and predators (e.g. Chapman & Bulleri, 2003); low species and genetic diversity (Johannesson & Warmoes, 1990; Chapman & Bulleri, 2003; Fauvelot et al., 2009); the dominance of flora and fauna that often represent an early stage of succession (Bacchiocchi & Airoldi, 2003; Bulleri & Airoldi, 2005; Glasby et al., 2006); and the facilitation of the settlement and spread of non-indigenous species (Bulleri & Airoldi, 2005; Moschella et al., 2005; Glasby et al., 2006; Vaselli et al., 2008). The effects of urban infrastructure and armouring can also scale up, causing alterations of coastal seascapes, create stepping stones or corridors for hard-bottom species and affecting the dispersion and connectivity in marine populations at regional scales (Glasby & Connell, 1999; Dethier et al., 2003; Airoldi et al., 2005). On one side increased connectivity could be a cost-effective way to enhance the conservation of threatened species and habitats, for example by providing new dispersal routes that facilitate their migrations in response to climate changes (Thomas, 2011). On the other, there could be also severe drawbacks, including the rapid expansion of "weedy" non native species that are particularly well adapted to these environments, as well as the breakage of natural distribution barriers among isolated (e.g. by stretches of sandy habitats) and differentially adapted populations (Fauvelot et al., 2009). For example, population genetic analysis on the limpet, *Patella caerulea*, from natural and artificial habitats at various sites along the Adriatic coast showed that genetic diversity (allelic richness and gene diversity) was significantly higher in populations inhabiting natural rocky shores than those on artificial structures (Fauvelot et al., 2009). While the causes of these differences are not yet understood and require further investigation, they clearly suggest that the expansion of armouring and other structures may lead to genetic diversity loss in rocky shore populations at regional scales.

Indeed, biotic homogenization is probably one of the major large scale impacts expected from increasing urbanization (Sax & Gaines, 2003).

Despite the impacts that have shown decreases in diversity of associated flora and fauna, at a first glance, coastal infrastructure and armouring seems to create suitable habitat for many marine organisms which rapidly settle and spread on the new hard substrata. It is precisely because of this trend that waste material is often dumped at sea to create artificial reefs. Indeed, it has been suggested that these artificial substrata may adequately represent natural habitats (e.g. Thompson et al., 2002; Pister, 2009) or may, in fact, compensate for loss of habitat elsewhere (e.g. Ianuzzi et al., 1996). Other authors have suggested adding more artificial structures to urban coastlines to create additional habitat (e.g. Iverson & Bannerot, 1984). This approach to conservation should, however, be treated with a great deal of caution without further research into the value of artificial substrata for survival of both common and rare species. There have been relatively few studies of the value of such reefs for species other than fish (see reviews by Baine, 2001; Svane & Petersen, 2001 and recent work by Perkol-Finkel et al., 2006; Miller et al., 2009; Burt et al., 2009). Similarly, there have been relatively few studies of the value of armouring and urban infrastructure as habitat for marine fauna and flora, although many subtidal epibiota are fouling species which rapidly colonize artificial structures (Glasby & Connell, 1999; Chapman & Clynick, 2006). Some structures, such as floating buoys and pontoons, create novel habitat for which there are no natural equivalents (Connell, 2000), whereas other surfaces, e.g. subtidal walls, may be closer in morphology to natural cliffs and rocky reefs and have similar biotic assemblages (Glasby, 1999). Although many species of fish aggregate around coastal infrastructure, such as marinas and wharves, these assemblages can be a reduced or different mix of species than occurs on natural reefs, depending on the type of habitat created by the artificial structures.

Effectiveness of artificial defence structures in a changing climate

But the concerns about human-made coastal defences are not just confined to the ecological issues. Their real efficiency is in question even from a management point of view. Indeed, the effectiveness of coastal defences is expected to be strongly reduced as consequence of the amplification of extreme weather events linked to climate change (Evans et al., 2004). Currently in Europe, large stretches of the coastline are retreating, and future scenarios are worsening (Zanuttigh, 2011). In fact, coastal areas are projected to be

exposed to increasing risks of erosion and land loss (IPCC, 2007). This phenomenon is related to the alteration of the frequencies and intensities of extreme weather events and rise in sea level, potentially linked with global climate change, which has mostly adverse effects on natural and human systems (IPCC, 2007). In effect, as described by Evans et al. (2004), higher sea levels will increase the frequency with which defence structures are overtopped by waves or very high tides. This increased overtopping will affect the inundation risk leading to failure of the structure itself. Further, storm surge would produce greater water depths at the structure and then larger waves exacerbating the overtopping. As a domino effect, larger waves would affect the stability of the structure with an increase of damage risk. In fact, the size of blocks that constitute the structure is directly proportional to the volume of the significant wave height. Therefore, every change in wave height at the defence structures can result in an increase in the size of the blocks required to achieve the same stability. Further, with larger waves at the structure, there is likely to be greater reflection from defence structures and increased scour of the beach at the structure's toe. The result is the increase of the potential for failure of the defence structure. Therefore, hard armouring could be cost-ineffective for the purpose of coastal protection (Evans et al., 2004).

The case of the United Kingdom is illustrative. The UK has a particularly long coastline that is subjected to erosion by the sea and the effects of the weather (Evans et al, 2004). Erosion can undermine flood defences or change the shoreline in ways that increase the risk of inland and coastal flooding. The predicted acceleration of rise in sea level under the climate scenarios will increase the effectiveness of coastal processes that operate at sea level and will have an important impact on erosion rates (Evans et al., 2004). Hence, it has been estimated that maintaining the existing defence structures and guaranteeing the actual protection standards will demand costs 1.5 to 4 times greater by 2080, depending on the emission scenario (Thorne et al., 2007). In other words, at present expenditure levels, approximately one-third of existing coastal defences in the UK could not be maintained in the future (Defra, 2001).

Alternative options for coastal protection

In some part of north Europe (e.g. United Kingdom, Netherlands), the focus in coastal protection management has been shifted to develop alternative options to hard armouring (Zanuttigh, 2011). For example, incorporating natural habitat (e.g. salt marsh, wetland

vegetation) that may provide a buffer against erosion in sheltered areas. In general the approach to coastal management is moving towards less aggressive flood defence and coastal protection. This approach favour environmental protection, with a preference for flood-management measures that have minimal environmental impact. Specifically, flood-management agencies in north Europe, are moving away from a perspective of 'flood defence' towards 'flood-risk management'. This means that land use planning and measures to reduce exposure and vulnerability to flood would be favoured over measures to reduce the physical hazard by defence structures. Such measures include managed retreat of defence structures (Rupp-Armstrong & Nicholls, 2007; Zanuttigh, 2011).

Managed retreat or realignment of hard coastal defence structures has been identified as an adaptive strategy for alleviating estuarine flood risk or for the re-establishment of ecologically valuable intertidal habitats, such as salt marshes and tidal flats (Townend & Pethick, 2002; Morris et al., 2004; French, 2008). Cost benefit analyses typically show a net advantage of managed realignment over other constructed defence options (Spurgeon, 1998; Turner et al., 2007). This involves dismantling or breaching shore defences and eliminating them or moving them inland, preferably taking advantage of natural topographic contours to reduce the cost of engineering to the standards required for alleviating flood risks (French, 2008; Townend, 2008).

Hence, the flood alleviation strategies include the ''no defend'' approach, that allow some areas to be sacrificially flooded. However despite the concerns about predicted increases in sea level and in the frequency of storm events, our understanding about the ecological impacts of flooding by seawater is relatively sparse. Indeed, with more frequent marine inundations, the sea level rises will potentially lead to an increase in salinity across the upper estuarine system, with consequent alterations of the transitional conditions (van der Wal & Pye, 2004; Wolters et al., 2005). Hence advances the knowledge of the environment vulnerability to seawater inundation is essential to proceed with alternative options, such as no-defence.

Scopes and organization of the thesis

My doctoral thesis analyses the consequences of different options for coastal protection, namely hard engineering 'artificial defences' (i.e. impact of human-made structures) and 'no-defence' (i.e. impact of seawater inundation), in different European areas (Fig. 1.1.). The thesis is structures around 4 research topics:

1)   The first step was to map and characterized the artificial structures built along the Sicilian coastline (Mediterranean sea). This study took in consideration not just the coastal-defence structures, since was focused to the overall artificialization of the coastline. The characterization is also aimed to evaluate the typology, distribution and extension of artificial structures to respect with the surrounding natural substrate. In fact, the role of the surrounding habitat on the ecological implication of artificial structures is poorly investigated (see Bulleri, 2005). This study is aimed to improve the general knowledge on the urbanization of the Sicilian coast and offer tools for further research on the effects structures are having on the marine environment.

2)   In this direction, was subsequently conducted a study on the fish assemblages inhabiting coastal-defence structures placed in different habitats along the Sicilian coastline, in order to experimentally test if the differences in assemblage composition among breakwaters and natural rocky reef would change depending on the nature of the surrounding habitat (sandy rather than rocky) of the structure. From one hand the largest body of literature regarding the ecological effects of artificial defence structures focused on intertidal and subtidal epibiotic assemblages (Bacchiocchi & Airoldi, 2003). So, I stressed the importance of investigate some ecological implication of artificial defence structures on the fish communities, given the economic and ecological role of coastal fishes (Horn et al., 1999). From the other hand, the studies about the differences in fish assemblages among artificial and natural reef not specifically focused the environmental context in which artificial structures are placed. In general the introduction of artificial defence structures in a prevalent sandy habitat causes the loss of soft-bottom habitats and affects the associated biological communities (Martin et al., 2005). Conversely artificial structures and adjacent natural rocky reefs are expected to offer similar structural features, and to be populated by benthic communities not significantly different (Perkol-Finkel et al., 2006). To explore the potential interactions between the artificial habitat and the surround environments in affecting the composition and structure of fish assemblages I conducted surveys along coastlines where artificial structures are deployed in both sandy and rocky context.

3)   A further aspect of the ecological impact of hard coastal-defence structures that I investigated, is that relative to the effect of the organic detritus detached from

Fig. 1.1. Map of Europe showing the areas interested by my doctoral research (red points). Photo modified from "Google Earth", 2013.

breakwaters on the macro-benthic organism living the natural sandy habitats. This study was conducted in the sandy shoreline of the north-west Adriatic sea, were artificial structures constitute the prevalent or only hard bottom. In this context, artificial infrastructures could have important consequences for the functioning of coastal system through the spatial subsidy of detrital material that is sloughed off the artificial structure and changes productivity, nutrient cycling, detrital pathways in the adjacent soft sediments.

The study consisted in identify and quantify the type, amount and distribution of detritus produced by dominant hard-bottom species on the structures, and then carried out a field manipulative experiment to assess the different effects of detritus from the dominant taxa (i.e. *Ulva* spp. vs *Mytilus galloprovincialis*) colonizing coastal defence structures on the adjacent soft–sediment macrofauna. I predicted different effects of these two forms of detritus because of the different nature in the organic material and the provision of shell material associated with *Mytilus* deposition. In this direction I analyzed the responses to detrital subsidy of both the macrofauna inhabiting the sediment under the detritus mat and the macrofauna directly associated to the detritus. In order to test whether any observed effects were consistent over time, responses were measured at different times within a short interval.

4)   Finally, I investigated the ecological implication of the "no-defend" option, which allow the inundation of coastal habitats. The focus of this study, conducted in

the south-west England, was the potential effect of seawater intrusion on the degradation process of marine, salt-marsh and terrestrial detritus, including changes on the breakdown rates and the associated macrofauna.

In Europe, but particularly in the UK, since the Great Flood, of January 1953 that hit the east coast of Great Britain, there has been invested a lot on maintaining and enhancing sea defences (Nicholls & De La Vega-Leinert, 2008). Nevertheless the high cost of maintaining this existing coastal flood-defence system, together with an increasingly holistic understanding of coastal processes, has led to the development of more sustainable strategies for the management of the coast (Zanuttigh, 2011). Integral to these new coastal management strategies is the appreciation of salt marshes as a key part of the coastal system (Rupp-Armstrong & Nicholls, 2007). Salt marshes are of immense environmental and economic importance, particularly for flood defence since their presence reduces the impact of waves at the shore line (Wolters et al., 2005). A salt-marsh restoration technique, called 'managed retreat', involves the tidal inundation of coastal land in front of re-aligned flood defences (Rupp-Armstrong & Nicholls, 2007).

Hence, the development of alternative strategies of artificial coastal defences includes the understanding of alterations in ecosystem functioning in a flooding scenario. Particularly in estuaries seawater incursion can differently affect habitats along a gradient from marine to freshwater. In rivers and estuaries the functioning of aquatic ecosystems is largely dependent leading by decomposition of allochthonous organic matter, enabling the recycling of nutrients and chemical elements, sustaining important food chains and primary production (Goñi et al., 2003). For this reasons, a thorough understanding of the potential changes of the detritus decomposition processes is critical. In this direction, I carried out a field experiment in which marine, salt-marsh and terrestrial detritus were exposed to degradation at three salinity regimes: seawater, brackish and freshwater.

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CHAPTER 2

Mapping and characterization of the artificial structures along the Sicilian coast

Introduction

Anthropogenic pressure on coastal habitats is increasing worldwide. Changes in demography and distribution of human populations have severe impacts on coastal landscapes. Today, many of the largest cities in the world are located in coastal zones and it is expected that two-third of the human population will live within 100 km of a coast by 2025 (EEA, 2006). Moreover coastal areas are affected by the addition of infrastructures needed to sustain residential, commercial and tourist activities. In the Mediterranean coast is predicted that seasonal tourism will reach 350 million people per year by 2025 (Hinrichsen, 1999).

Nowadays, in Europe 22000 km^2 of the coastal zone are covered in concrete or asphalt, and about 50 % of the Mediterranean shorelines bordering Spain, France and Italy are dominated by artificial structures (more than 1500 km), of which most are developed for harbours and ports (Airoldi & Beck, 2007 and references therein). Moreover, since large stretches of European coasts are already retreating and projected scenarios are worsening, many artificial structures, such as breakwaters and seawalls, are built as tool against coastal erosion (Zanuttigh, 2011).

Despite artificial structures produce widespread changes that alter the coastal zones by causing the loss and fragmentation of natural habitats, the ecological consequences of their introduction into shallow coastal waters have received relatively little attention (Southward & Orton, 1954; Glasby & Connell, 1999; Hawkins et al., 2002; Chapman, 2003; Bulleri, 2006; Di Franco et al., 2011).

The aim of this study is to map and characterize the extent and typology of urban marine infrastructures along the coast of Sicily (Italy), in order to improve the general knowledge on the urbanization of the Sicilian coast and offer tools for further research on the related ecological issues. Mapping the spatial distribution of different types of coastal structures categorised according to their physical characteristics is, in fact, a first step towards understanding and predicting the effects structures are having on the marine environment.

The island of Sicily (Mediterranean sea) is one of the most populated regions in Italy with more than 5 million residents. Most of the human population lives in cities located along the coast (Palermo, Catania, Messina, Trapani, Siracusa, Agrigento). Hence, large commercial ports, industrial facilities and several structures for coastal protection border the coast of Sicily. Furthermore in the past decades many houses, railways, roads were built near shore. Despite the proliferation of urban marine structures, very few studies have focused on the changes introduced by these structures in the coastal environment, and have only covered very limited local spatial scales (Anfuso & Martínez del Pozo, 2005). As in other regions, there is a lack of comprehensive inventory and monitoring of their extension, typologies, and distribution, which would be the first step towards the development of an integrated management plan for increasingly urbanised coastal areas.

In the absence of a scientific definition of "urban marine environment" (Bulleri, 2006), I considered as urban those traits of the coast where any anthropogenic structure or activity affect the morphology of the coastline preventing its natural evolution. A database was constructed using information from Google Earth and structures were categorises based on their main physical attributes (typology, linear extension) and prevalent surrounding substrata.

Methodology

Study area

The Sicilian coasts extend for 1152 km, excluding the smaller islands, and have a considerable variability of environmental and infrastructure on the three main sides (Assessorato Territorio e Ambiente, Regione Sicilia, 2002).

The North side (Tyrrhenian sea; from "Capo Lilibeo" at west to "Capo Peloro at east) is characterized by an alternation of low coasts and flood plains often subject to accelerated erosion.

The East side (Ionian sea; from "Capo Peloro" at north to "Capo Passero" at south) is characterized, moving from north to south, by a succession of flood plains and coastal terraces. Further south, at the Gulf of Catania, opens a vast alluvial plain. In the most southern, alternate different morphotypes, including mountains, narrow beaches, and the beaches characterized by marshes and coastlines to high terraces.

The South side (Strait of Sicily and Strait of Malta, from "Capo Passero" at east to "Capo Lilibeo" at west) presents narrow stretches of beaches, bounded on the inside by the hills.

This trait of coast has low rocky coast marine terraces set with beaches located at the foot of the coastal cliffs, subject to intense erosion.

Mapping procedure

The study consisted into mapping and classifying every artificial structure built along the Sicilian coastline using satellite images and building a database including the main information on each structure. All the measurements and structure characteristics were derived using images and tools provided by Google Earth. Google Earth has become a useful mapping tool and is considered accurate when measuring distances across low-relief surfaces such along many Mediterranean coastlines (Nicolas et al., 2010; Harris et al., 2011; Waltham & Connolly, 2011). The total linear extent was measured using the scale ruler function in Google Earth (Waltham & Connolly, 2011).

As a first step, the location of any artificial structure with respect to the coastal side (North, East, South) was recorded. Subsequently, the linear length of the trait of shores interested by the urban structure was measured. The extension of the submerged part of the artificial structure and infrastructure or, in the case of defence barriers, the trait of shore interested to the protection were considered. Finally, the artificial structures were classified into the following categories:

- DS - Defence structures: includes docks, groynes, and seawalls for the defence of the roads or railway lines, barriers for the protection of houses next to the coastline, breakwaters of ports and harbours.
- DB - Detached breakwaters: breakwaters for coastal protection against erosion.
- PI - Ports and industrial plants: includes commercial ports, marinas, and traits of coast in front of industrial plants such as petrochemical Priolo, Gela and Milazzo.
- OS - Other structures: includes all the smaller artificial structures that do not play a role in the coastal defence. For example, small jetties, boat ramps, piers for the mooring of small boats, beach facilities.

This information was organized into a database also including information on the prevailing surrounding coastal substrate: rocky, sandy, artificial, or, in case of coexistence, rocky/sandy, rocky/artificial, sandy/artificial.

The dates of aerial images used in Google Earth were the most up-to-date available at by the period April-July 2011, although there were also examined, older images to overcome situations where the details of systems were obscured by cloud cover.

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The measurements and the characterization of the above parameters were done using images and tools provided by Google Earth.

Google Earth has become a useful mapping tool and is considered accurate when measuring distances across low-relief surfaces such along many Mediterranean coastlines (Nicolas et al., 2010; Harris et al., 2011; Waltham & Connolly, 2011).

The total linear extent was measured using the scale ruler function in Google Earth (Waltham & Connolly, 2011).

Unfortunately, the use of satellite images entail limits. In first instance for the identifications on a better details of the types of surrounding environment. It was not possible, for example to distinguish among the different nature of rocky or sandy substrates, as well as no information are available about the slopes of the coast. Moreover, minor structures, often classified under the category "Other structures", might be underestimated.
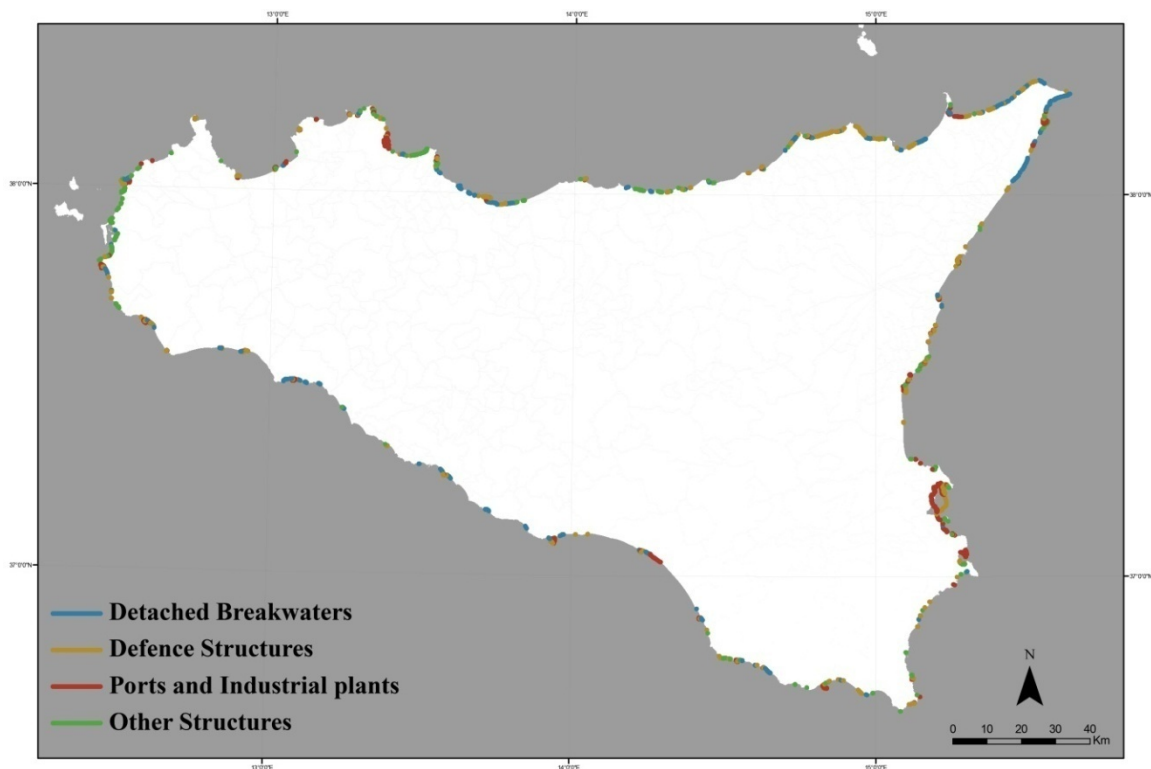

Fig. 2.1. GIS map visualizing the distribution of each type of artificial structure along the Sicilian coastline.

Distributon of urbanization
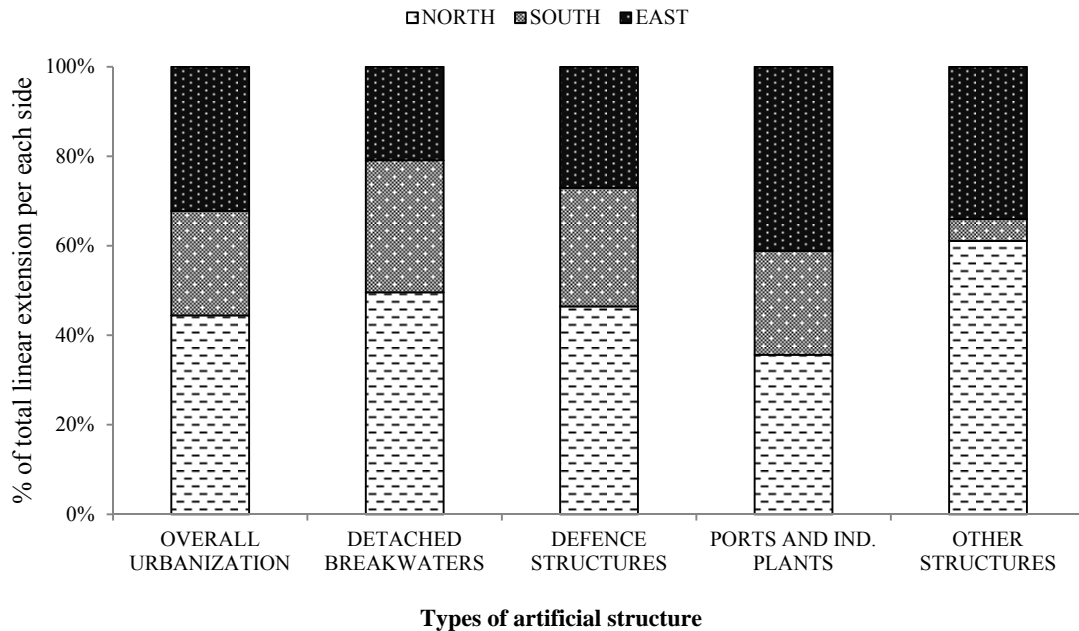
☐NORTH ▨SOUTH ■EAST

Fig. 2.2. Histogram showing the percentage distribution of each type of artificial structure all over the Sicilian coastline and at each coastal side.

Results

I identified and classified 583 sites, distributed 263 in the North, 136 in the South and 184 in the East (Fig. 2.1.). Along about 1152 km of coastline, more than 331 km are urbanized or artificial stabilized. However it is worth to highlight that for the measurement of the urbanized coast I considered the surfaces of the inner perimeter of the ports, including the inner and outer sides of the port's breakwaters. Instead, the calculation of the total extent of the Sicilian coastline was made by considering just the profile of the coasts (Assessorato Territorio e Ambiente, Regione Sicilia, 2002).

Overall infrastructures associated to ports and industrial plants account for the 40,15 % of the urbanized coast, followed by coastal defence structures (27,70 %) and defence barriers (20,68 %). Small structures classified as "Other structures" comprise only 11,45 % of the mapped infrastructures (Table 2.1).

Extent of artificial structures at each side
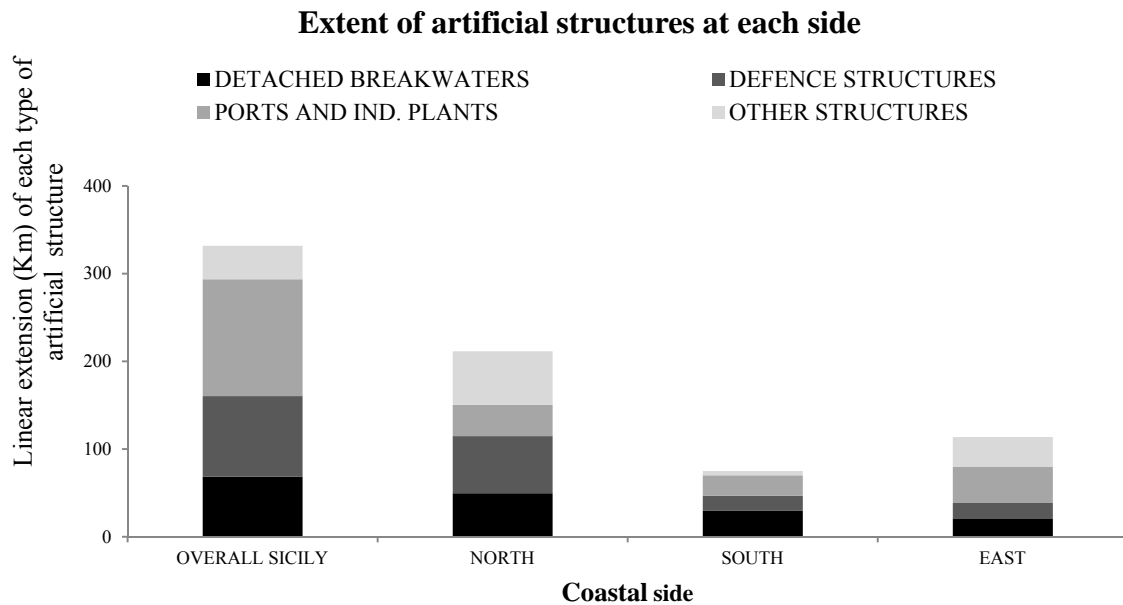
Fig. 2.3. Histogram showing the linear extension (km) of each type of artificial structures along the Sicilian coastline and along each of the coastal sides (north, east and south).

Substrate surroinding artificial structures
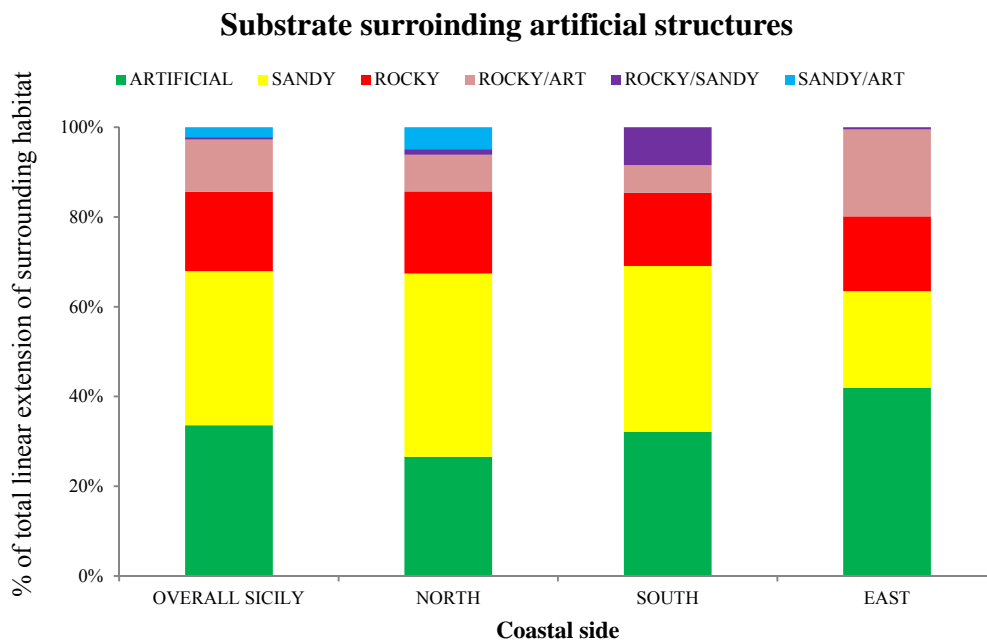
Fig. 2.4. Histogram showing the % of the substrates surrounding the artificial structures all over the Sicilian coastline and at each coastal side.

Looking at the details of each type of artificial structures along the three coastal regions of Sicily, it is clear that the Defence structures are developed mainly in the North coast (46,4 %), as well as Detached breakwaters (49,6 %) and Other structures (61,1 %). Instead, Ports

and industrial plants are developed prevailingly along the East side (Ionian sea) (Fig. 2.2. and Fig. 2.3.) where they comprise 41,1 % of the structures present.

Regarding the substrates surrounding the artificial structures, overall the dominant types are sandy (34,3 %) and artificial (33,6 %), whereas rocky accounted for the 17,7 % (Fig. 2.4.). This trend is quite consistent along North and South sides. While in the East side, artificial substrata is the dominant type. This latter result is driven by the big port of Augusta and the Petrochemical of Priolo, both located on the East side.

Within the categories of defence structures and detached breakwaters (Fig. 2.5.), the relative percentage of rocky substrata increase, especially in the South and in the East (Fig. 2.6.).
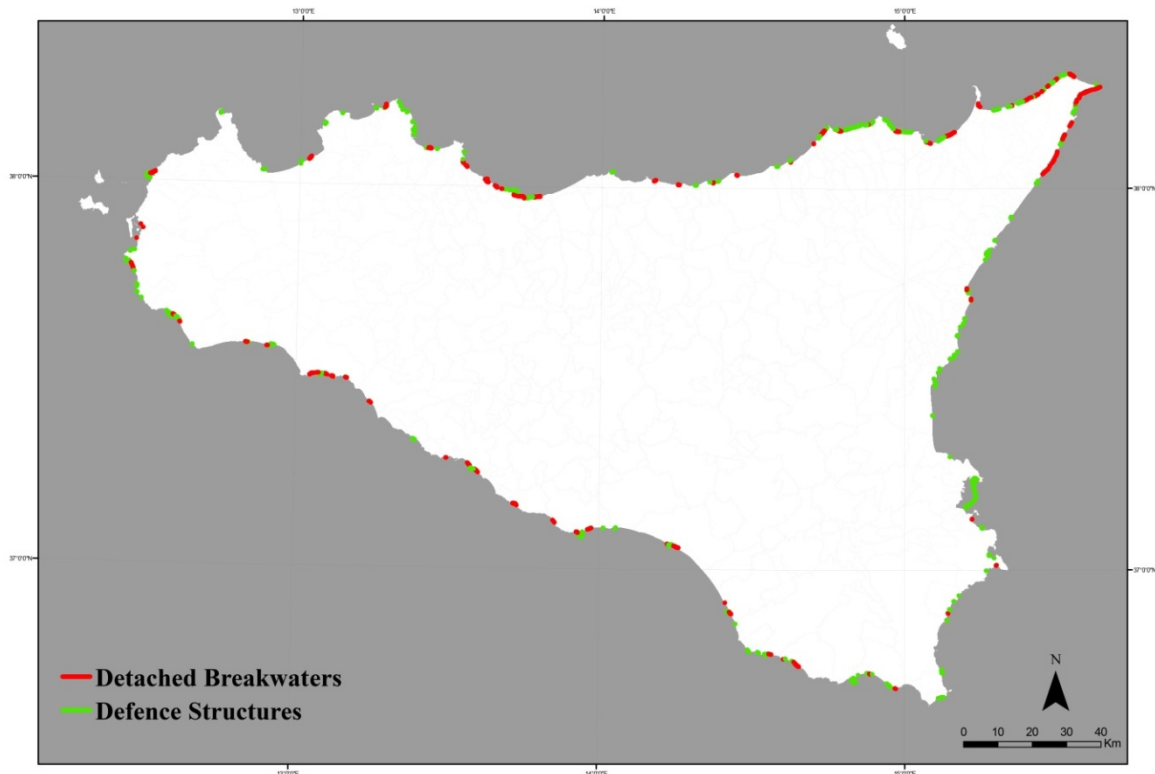
Fig. 2.5. GIS map visualizing the distribution of detached breakwater and defence structures along the Sicilian coastline.

The detached breakwaters are located prevailingly along sandy shores (with 74 % of the breakwaters built in sedimentary habitats), while the remaining 26 % is distributed between rocky (12 %), sandy/artificial (8 %) and rocky/sandy (6 %) (Fig. 2.7).

Comparing the extension of detached breakwaters on rocky and on sandy coast, is evident by the fig. 2.7. that the dominant substrate surrounding the barriers in each of the three

sides is the sandy one. The barriers along rocky shore are mainly developed along the North and secondly along the South side, whereas on the East they are rare (Fig. 8).

SUBSTRATES SURROINDING DB & DS

Fig. 2.6. Histogram showing the % of the total linear extension of the substrates surrounding the detached breakwaters and defence structures all over the Sicilian coastline and at each coastal side.

Substrates surrounding detached breakwaters

Fig. 2.7. Circle graph showing the % of the total linear extension of substrates surrounding the detached breakwaters all over the Sicilian coastline.

Detached breakwaters on rocks and on sand

■ SANDY ■ ROCKY

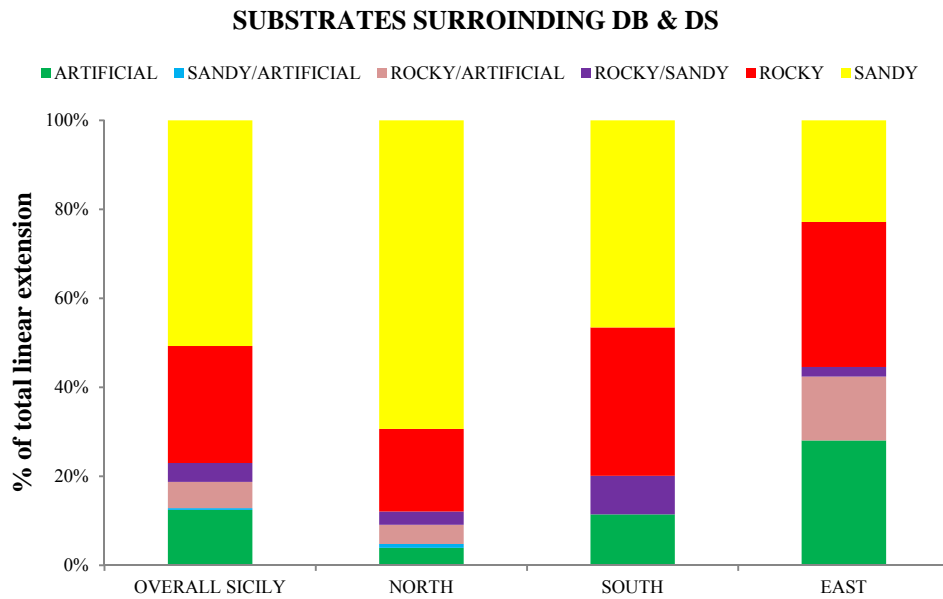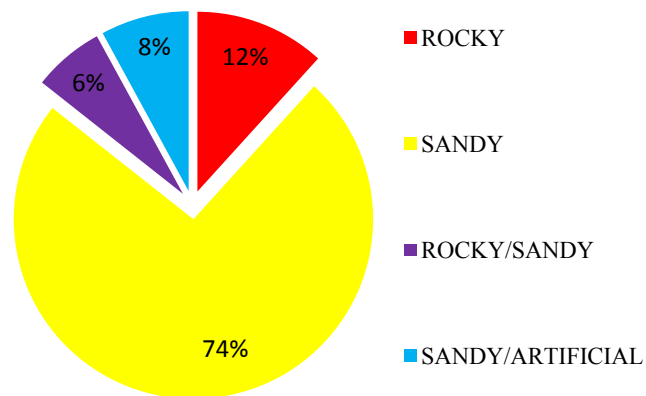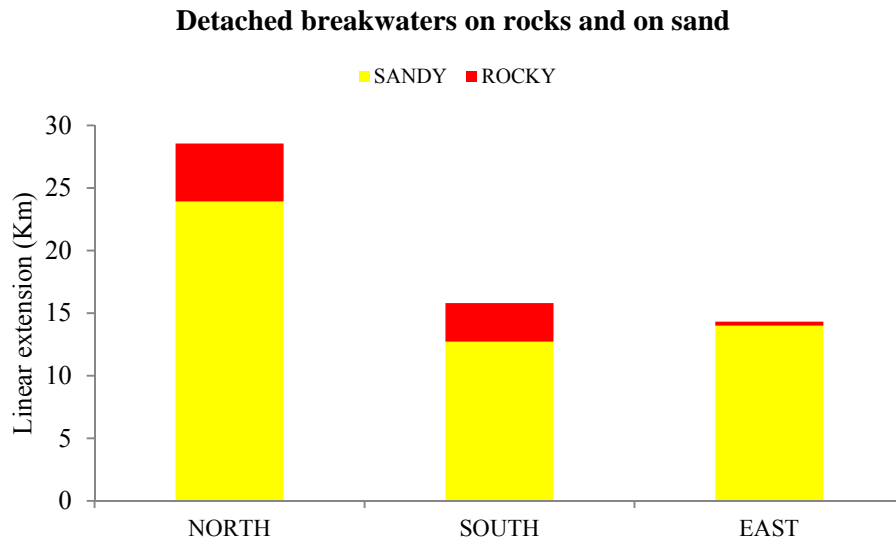**Fig. 2.8.** Histogram showing the % of the total linear extension of the sandy and rocky substrates surrounding the detached breakwaters at each coastal side.

Discussion

This study quantified and characterized, for the first time, the urbanization along the coast of Sicily. More than 331 km of coasts resulted artificial stabilized, protected or heavy urbanized. The greater portion of this coastal urbanization is related to the presence of ports and industrial plants, among which petrochemicals. They cover an important portion of coastline, in particular along the Ionian side, because the presence of the heavy urbanized area of Augusta and Priolo, near the city of Siracusa. The presence of ports in Sicily is relevant, if we consider that Sicily hosts the higher number of ports among the Italian regions, accounting for 43 ports on a total of 179 distributed along the national territory (Geoportale Nazionale, Ministero dell'Ambiente e della Tutela del Territorio e del Mare).

Moreover many breakwaters and seawalls are built to defend the coast from erosion. In fact, 33 % of the Sicilian coast (373 km) is actively retreating (Geoportale Nazionale, Ministero dell'Ambiente e della Tutela del Territorio e del Mare). The structures for coastal defence are abundant and widespread all over the Sicilian coastline, with a major presence in the Tyrrhenian side. These anthropogenic structures have become ubiquitous globally, causing large scale landscape modifications as well as changes of hydrodynamic and physical conditions with consequent impacts on the biological communities in shallow

coastal waters (Bulleri & Chapman, 2010). However, these impacts may vary according to the nature of the surrounding habitat of artificial infrastructures (Bulleri, 2005).

In this study I described the distribution of detached breakwaters in relation to the surrounding natural habitat. Detached breakwaters resulted to be prevalently displaced on sandy bottoms. On the other hand the breakwaters on a rocky habitat are moderately extended, specifically along the north and the south sides whereas in the east they are rare.

The breakwaters are mainly distributed on sandy coasts since they are the most common artificial constructions preventing or reducing shoreline erosion along European coasts. However those structures built over soft-bottom has multiple ecological implications (Bulleri, 2005). There is considerable evidence that by interrupting wave action, defence structures modify the nearshore water circulation, leading to changes in bottom topography, sediment grain size and organic content (Airoldi et al., 2005; Martin et al., 2005; Bertasi et al., 2007). Breakwaters also influence species abundance, distribution patterns and community structure of fauna from adjacent soft bottoms (Bertasi et al., 2007; Colosio et al., 2007). Furthermore in prevalently sandy coastlines, artificial structures offer novel hard-surfaces for the settlement of intertidal and subtidal rocky species otherwise absent in the area (Airoldi et al., 2005).

Furthermore, introducing artificial surfaces onto rocky bottoms is sometimes considered not to alter the fundamental nature of the habitat, especially when these structures are built with natural stones. It has, in fact, been assumed that the structure and functioning of assemblages that colonize those surfaces are analogous to those living on adjacent natural rocky shores (Thompson et al., 2002). However, some studies found that epibiota living on and fish assemblages associated with artificial structures differ from those on natural reefs (Glasby & Connell, 1999; Rilov & Benayahu 2000; Perkol-Finkel & Benayahu, 2004; Moschella et al. 2005; Clynick et al., 2008).

Generally the assemblages inhabiting the artificial structures are often characterized by a low species diversity compared with natural habitats (Bacchiocchi & Airoldi 2003; Chapman, 2003; Martin et al., 2005). The decrease in species diversity weakens the community resistance to abiotic and biotic disturbances, favouring the establishment of invasive species (Elton, 1958). In fact, artificial structures play a key role for the introduction and spread of invasive species (Bulleri & Airoldi, 2005; Glasby et al., 2006; Vaselli et al., 2008; Dafforn et al., 2012).

On a regional scale, artificial structures can function as corridors or stepping stones (Glasby & Connell, 1999), connecting otherwise separated populations. This phenomenon

promotes the homogenization of biota which is currently considered as a major anthropogenic impact (McKinney & Lockwood, 1999). In fact urbanization is today known as one of the leading causes of species extinction and biodiversity loss (McKinney, 2006).

The implications of the ongoing coastal urbanization require carrying out more ecological studies that, by incorporating rigorously designed field experiments, offer knowledge to guide future management practices.

Table 2.1. Summary of the extension (km) and number of each type of artificial structure along the north, south, east sides of Sicily and along the total Sicilian coastline; extension (km) and number of artificial structures in relation of the substrate were they are placed, along the main coastal sides and the total Sicilian coastline; extension (km) and number of detached breakwaters in relation to the substrate were they are placed, along the main coastal sides and along the total Sicilian coastline.

	NORTH		SOUTH		EAST		OVERALL	
Distribution of artificial structures	km	n°	km	n°	km	n°	km	n°
DETACHED BREAKWATERS	34,0	45	20,2	32	14,3	31	68,6	108
DEFENCE STRUCTURES	42,7	80	24,3	61	24,9	63	91,9	204
PORTS AND IND. PLANTS	47,5	28	30,9	21	54,8	36	133,2	85
OTHER STRUCTURES	23,2	110	1,9	22	12,9	54	38,0	186
TOTAL	**147,4**	**263**	**77,4**	**136**	**106,9**	**184**	**331,7**	**583**
Urbanization and substrates								
ARTIFICIAL	39,1	29	24,8	22	46,1	18	110,1	69
SANDY	60,2	116	28,6	57	23,7	60	112,5	233
ROCKY	27,0	80	12,6	38	18,3	82	57,9	200
ROCKY_ARTIFICIAL	12,0	24	4,8	6	21,3	24	38,1	54
ROCKY_SANDY	1,8	6	6,5	13	0,5	1	8,9	20
SANDY_ARTIFICIAL	7,2	6	0,0	0	0,0	0	7,2	6
Detached breakwaters and substrates								
DB ON ROCKY	4,6	8	3,1	7	0,3	2	8,0	17
DB ON SANDY	23,9	33	12,7	19,0	14,0	29	50,6	81
DB ON SANDY/ART	5,4	4	0,0	0,0	0,0	0	5,4	4
DB ON ROCKY/SANDY	0,0	0	4,4	6,0	0,0	0	4,4	6

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CHAPTER 3

Fish assemblages associated to artificial defence structures and relationships with the surrounding environmental context

Introduction

Erosion and flooding are common phenomenon along the coastlines worldwide, exacerbated by the sea-level rise and the increase of storm frequency potentially related to the global climate change (National Institute of Coastal and Marine Management of the Netherlands, 2004). In Europe, where more than 15000 km of coast are currently retreating, numerous artificial coastal-defence structures (i.e. breakwaters and seawalls) have been built over the past decades as measure for coastal protection (Airoldi & Beck, 2007).

The introduction of artificial defence structures causes ecological impacts at local and regional scale (Airoldi et al., 2005), because they modify both physical and biological features of the natural environment. The largest body of literature regarding the ecological effects of artificial defence structures focused on intertidal and subtidal epibiotic assemblages (Bacchiocchi & Airoldi, 2003). In most instances infrastructure and armouring structures are built in areas that would otherwise be sedimentary, thereby introducing new intertidal or subtidal hard substrata where that was not previously available (Dethier et al., 2003; Airoldi et al., 2005). Despite the impacts that have shown decreases in diversity of associated flora and fauna, at a first glance, coastal infrastructure and armouring seems to create suitable habitat for many marine organisms. In fact, sessile communities associated with hard substrata respond quite clearly to the presence of the artificial hard surfaces rapidly settling and spreading. This results in the introduction of new species to these areas, with consequent local alteration of species composition, abundance and diversity (Airoldi et al., 2005).

Many species of fish aggregate around coastal infrastructure, such as marinas and wharves. These assemblages can be a reduced or different mix of species than occurs on natural reefs, depending on the type of habitat created by the artificial structures (e.g. Rilov & Benayahu, 1998; Cenci et al., 2011; Pizzolon et al., 2008; Santin & Willis, 2007; Clynick, 2006; Guidetti et al., 2005; Guidetti, 2004). The interest on how fishes respond to the presence of artificial defence structures is not surprising considering that fish communities

in the shallow coastal waters are economically and ecologically important (Horn et al., 1999) but also highly threatened by habitat alteration (Greene & Shenker, 1993; Bussotti et al., 2003; Guidetti, 2004; Guidetti et al., 2005). Studies carried out along the Italian coasts found strong differences in composition among fish assemblages inhabiting respectively artificial defence structures and sandy bottoms (Guidetti, 2004), but not differences were detected between breakwaters and nearby rocky reefs (Clynick, 2006). On the contrary, in a study conducted in Taiwan, Wen at al. (2010) found different fish assemblages between breakwaters and nearby rocky reefs.

Despite some findings suggest that the habitat surrounding artificial coastal-defence structures influences the degree of the their effect on biological assemblages, just few studies tackled this issue (Bulleri, 2005). In general the introduction of artificial defence structures in a prevalent sandy habitat causes the loss of soft-bottom habitats and affects the associated biological communities (Martin et al., 2005). Conversely artificial structures and adjacent natural rocky reefs are expected to offer similar structural features, and to be populated by benthic communities not significantly different (Perkol-Finkel et al., 2006).

To explore the potential interactions between the artificial habitat and the surround environments in affecting the composition and structure of fish assemblages I conducted surveys along coastlines where artificial structures are deployed in both sedimentary and rocky context.

The specific aim of the study is to test: 1) whether the composition and structure of fish assemblages differ between artificial structures and natural reefs, and 2) if the composition and structure of fish assemblages inhabiting artificial structures differs between structures located along sandy *vs* rocky coastlines.

Materials & methods

Study area

This study was carried out at two different locations, each extending about 10 to 20 km along the coast of Sicily (Fig. 3.1.). Those locations were selected among those traits of coast where artificial defence structures are deployed in both sandy and rocky habitat, and where natural references reef sites are available. In order to achieve this purpose a preliminary census of every artificial structure all over the Sicilian coastline in relation to its surrounding environment was a crucial prerequisite (see Chapter 2 of this thesis).
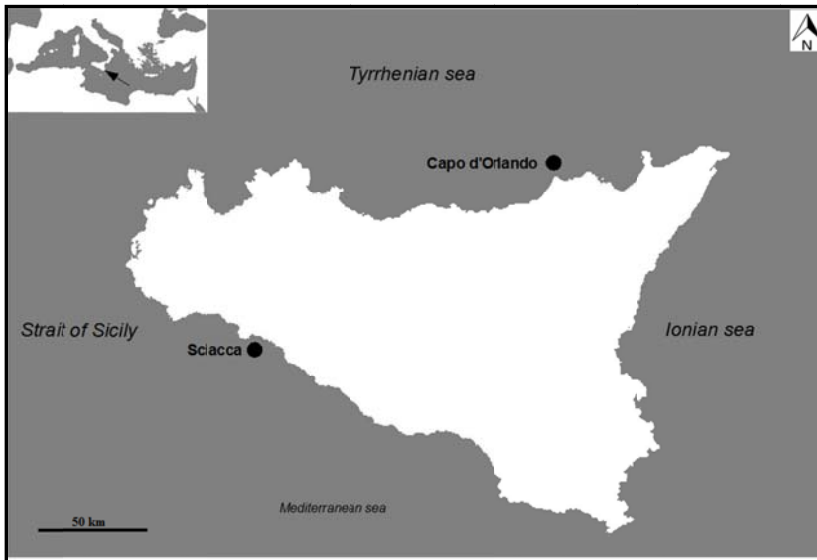
Fig. 3.1. Map of the study area and indication of the two locations where the surveys were carried out.

The two locations included in the study were: Sciacca (hereafter 'SCI') along the south-western coast of Sicily; 37°30'11.15"N, 13°04'44.33"E; and Capo d'Orlando (hereafter 'CDO') in the southern Tyrrhenian sea, on north-eastern Sicily; 38°09'59.26"N, 14°45'01.04"E. Both locations display short narrow beaches, surrounded by mountains or hills, which interrupt the continuity and are subjected to erosion. Variability, at a 'coastline' spatial scale (tens of kilometres), is largely influenced by local factors such as coastal geology, currents and wave regimes and riverine runoff (Airoldi et al., 2005). For this reason I used a broader geographical scale (hundreds of kilometres), in order to provide results not affected by specific local conditions and then generalizable.

In each location surveys were conducted in three habitats: artificial defence structures in a prevalent sandy coast (AS), artificial defence structures in a prevalent rocky coast (AR), and natural rocky reef (NR). At each location there were selected stretches of coast (extending hundreds of meters), characterized continuously by the prevalence of one of the three habitat types, hereafter denominated "sites".

In each location nine sites were selected haphazardly, separated each other from hundreds of meters to kilometres. At 'CDO' there were three sites for each of the habitat type: (NR, AS, AR). At 'SCI', since only in two sites the conditions for AS category were satisfied, there was selected an unbalanced number of sites per habitat. At SCI location, therefore, 5 sites were classified as AR, 2 sites as AS and 3 as NR (Fig. 3.2).
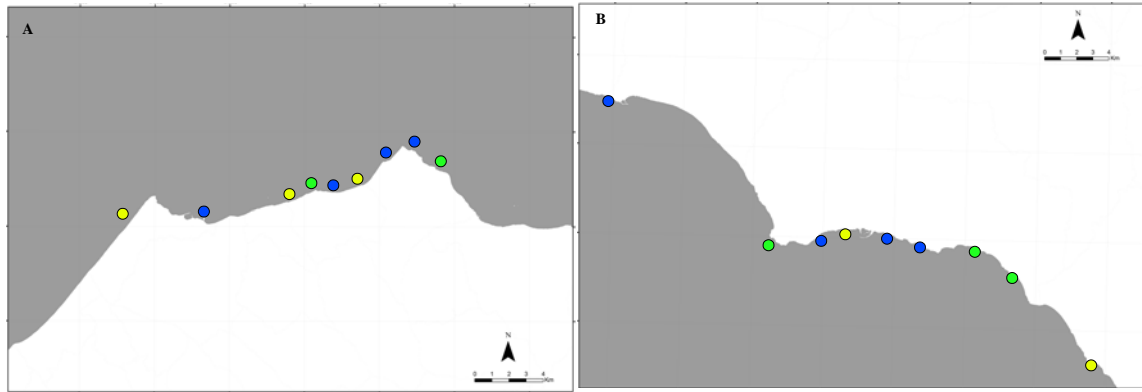
Fig. 3.2. Maps with localization of sampling sites at A) Capo d'Orlando and at B) Sciacca. Blue = AR (Artificial structure on rocky habitat); Green = NR (Natural rocky reef); Yellow = AS (Artificial structure on sandy habitat).

Sampling methodology

Data were collected by non-destructive *visual census* method along a strip-transect of 25 m long and 2 m width. Transect's width was modified from the one usually adopted in rocky habitat (5 m, Harmelin-Vivien et al., 1985) and adapted to the width of the seaward side of the artificial structures often not exceeding 4 meters (authors, personal observation). The fish counts was performed during the laying of the transect line, in order to limit the operator effect on fish behaviour (Kulbicki, 1998; Edgar et al., 2004; Dickens et al., 2011). Sampling was carried out in August 2011 within two weeks to minimize temporal differences. All counts were performed under good meteorological conditions within a depth range of 0-3 m and between 9.00 and 15.00 h to minimize the diurnal variability between transects (Willis et al., 2006). Given the shallow depth the survey was carried out by snorkeling to evaluate species composition and abundance.

Eight replicate transects were performed at each site by three divers that were previously trained to standardize sampling procedures. To reduce any potential bias in surveying difference groups of fish (i.e. pelagic and benthic) two censuses were performed along the same transect, one at faster speed (approx. 6m/minute) to census mobile fishes and the second at lower speed (approx. 3.5m/minute) to census benthic and crypto benthic species (De Girolamo & Mazzoldi, 2001).

According to Guidetti et al. (2002) and Pais et al. (2007), fish taxa with similar ecological features were aggregated into seven ecological categories (EC; in some cases corresponding to families), defined on the basis of their feeding habits and spatial organization within the water column. (1) BEN: benthic meso-carnivorous fishes (e.g.

Blenniidae, Gobiidae, *Mullus* and Tripterygiidae); (2) HER: herbivorous fishes (i.e. the sparid *Sarpa salpa*); (3) LAB: mesocarnivorous nectobenthic fishes belonging to the Labridae family; (4) PLA: planktivorous fishes inhabiting the water column, often aggregated in schools (e.g. Atherinidae, Pomacentridae and the sparid *Oblada melanura*); (5) POM: particulate organic matter feeders (i.e. Mugilidae); (6) SER: site-attached piscivorous fishes belonging to the Serranidae family; (7) SPA: meso and macrocarnivorous sparids belonging to the genus *Diplodus* and *Sparus aurata* (Table 3.1.).

Data analyses

The putative differences between the assemblage's structure and composition were analysed using a factorial design. The model included the main effects of Habitat (HA, fixed) and Location (LO, random) and Site (SI, random) that were nested in the interaction (LO x HA).

Statistical analyses were carried out on both multivariate and univariate data sets. Whole assemblage structures (abundance data) were analysed using a three-way permutational multivariate analysis of variance (PERMANOVA; Anderson, 2001; Anderson et al., 2008), according to the design reported above. In order to reduce the weighting of abundant species (e.g. those forming large schools) and increase that of rare taxa, data were log (x + 1)-transformed.

The Similarity Percentages procedure (SIMPER) was used to identify the species mostly contributing to the dissimilarity between habitats and only those variables whose contribution exceeded an arbitrary chosen threshold value of percentage dissimilarity >5% were shown.

In addition, a univariate PERMANOVA based on Euclidean distance was used to investigate patterns of distribution of total abundance of fish and species richness, avoiding any assumption about the distribution of the variable.

Significance was set at p = 0.05, p-values being obtained using 9999 permutations of residuals under a reduced model (Anderson, 2001). Non-metric Multidimensional Scaling (nMDS) (Clarke, 1993) based on the Bray-Curtis similarity matrix was used to visualize the ordination of samples within a two-dimensional space. As there were too many observations to view in a single ordination (totally n = 144), there were examined the eighteen centroids for the combined factor Habitat-Location.

All the statistical analyses were performed with PRIMER 6 + PERMANOVA software package from Plymouth Marine Laboratory, UK (Clarke & Warwick, 2001; Anderson et al., 2008).

Table 3.1. List of fish taxa (+: present; 0: absent) and ecological categories (see materials and methods) recorded at each of the two locations on artificial structures on rocky environment (AR), Natural reef (NR) and artificial structures on sandy environment (AS). Ecological categories of fishes: HER: herbivorous; PLA: planktivorous; BEN: benthic; LAB: labrids; POM: particulate organic matter feeders; SER: serranids; SPA: sparids.

| Family | | Capo d'Orlando | | | | | | | | | Sciacca | | | | | | | | |
| | | AR | | | NR | | | AS | | | AR | | | | NR | | | AS | |
Species	EC	1	2	3	1	2	3	1	2	3	1	2	3	4	1	2	3	1	2
Atherinidae																			
Atherina sp.	PLA	+	+	+	+	+	+	+	+	+	0	+	0	0	0	0	+	0	0
Blenniidae																			
Aidablennius sphynx	BEN	0	0	+	0	+	+	+	0	+	0	0	0	0	0	0	+	0	+
Parablennius gattoruggine	BEN	0	+	0	+	0	0	0	+	0	0	0	0	0	0	+	0	0	+
Parablennius sanguinolentus	BEN	0	+	+	0	+	+	+	+	+	0	0	0	0	0	0	+	0	+
Carangidae																			
Trachinutus ovatus		0	0	0	0	+	+	0	0	+	0	0	0	0	0	0	0	0	0
Labridae																			
Coris julis	LAB	+	+	+	+	+	+	0	+	+	+	+	+	+	+	+	+	+	+
Labrus merula	LAB	0	0	0	0	0	0	0	0	0	0	+	+	+	0	+	0	0	0
Labrus viridis	LAB	0	0	0	0	0	0	0	0	0	+	0	0	0	+	+	0	0	0
Symphodus ocellatus	LAB	+	+	0	+	0	0	0	+	0	+	+	+	+	+	+	+	+	+
Symphodus roissali	LAB	+	+	0	+	0	0	0	+	+	+	+	+	+	+	+	+	+	+
Symphodus tinca	LAB	+	+	+	+	0	+	+	+	+	+	+	+	+	+	+	+	+	+
Thalassoma pavo	LAB	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+
Moronidae																			
Dicentrarchus labrax		0	+	0	0	0	0	+	+	+	0	0	0	0	0	+	0	+	+
Dicentrarchus puntactus		0	0	0	0	0	0	0	0	0	0	0	0	0	0	+	0	+	+
Mugilidae																			
Mugil sp.	POM	+	+	0	+	+	+	+	+	+	+	+	+	0	+	0	+	+	0
Mullidae																			
Mullus surmuletus	BEN	+	+	0	+	0	0	+	0	+	+	+	+	+	+	+	+	+	+
Pomatocentridae																			
Chromis chromis	PLA	+	+	+	+	+	+	0	+	+	+	+	+	+	+	+	+	+	0
Serranidae																			
Serranus cabrilla	SER	+	0	0	0	0	0	0	0	0	+	+	0	0	0	0	0	0	0
Serranus scriba	SER	+	+	+	+	+	+	0	+	+	+	0	+	+	+	+	0	0	0
Sparidae																			
Diplodus annularis	SPA	0	0	0	0	0	0	+	0	0	0	+	+	+	0	0	+	+	0
Diplodus sargus	SPA	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+
Diplodus puntazzo	SPA	+	+	0	+	0	0	+	+	+	0	0	+	0	0	0	0	+	0
Diplodus vulgaris	SPA	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+
Lithognathus mormyrus	SPA	+	0	0	0	0	0	+	+	0	0	0	0	0	0	0	0	0	0
Oblada melanura	PLA	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+
Sarpa salpa	HER	0	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+	+
Tripterygiidae																			
Tripterygion tripteronotus	BEN	0	0	0	0	0	0	+	0	0	+	+	0	0	+	+	0	0	0

Results

General description

Apart from the cases of Atherinidae and Mugilidae for which visual identification was possible only at family level, the remaining taxa were identified at genus or species level.

A total of 28 fish taxa belonging to 11 families were recorded during the surveys. In the location 'CDO' the number of taxa recorded was 24, in 'SCI' was 26, whereas the taxa in common to both locations were 22. The whole fish assemblage was dominated by *Atherina* sp., *Chromis chromis*, *Coris julis*, *Diplodus sargus*, *Oblada melanura*, *Mullus surmuletus*, *Sarpa salpa*, *Symphodus ocellatus*, *Symphodus roissali* and *Thalassoma pavo* (Fig. 3.3.).

Overall, 9308 individuals were recorded. Among the ecological categories, PLA represent the 32,4 % of the total number of individuals, HER the 23,3 %, LAB the 22,4 % and SPA the 11,6 % (Fig. 3.4).

Fish assemblage structure

The nMDS plot showed segregation among locations and high variability among sites (Fig. 3.5.). The variability between sites was pronounced for the location 'CDO' than 'SCI' where the points in the plot were distributed quite close each other. By contrast it is not evident a clear trend among habitats.

The multivariate analyses performed on the fish assemblage revealed significant differences among the investigated habitats (*pseudo*-F = 2,697; p = 0,015), locations (*pseudo*-F = 7,513; p = 0,0001) and sites (*pseudo*-F = 4,096; p = 0,0001) (Table 3.2). Pairwise comparisons for the term "Habitat" revealed that fishes inhabiting AS differed from fishes populating NR (t = 2,344; p = 0,0129). No differences were detected between AS *versus* AR (t = 1,6049; p = 0,0837) and between AR *versus* NR (t = 1,2957; p = 0,2418) (Table 3.2.).

The SIMPER showed the 7 fish taxa individually contributed more than 5 % to the dissimilarities between AS and NR. *Sarpa salpa*, *Oblada melanura*, *Mullus surmuletus* and *Thalassoma pavo* resulted more abundant in AS relative to NR. In contrast, *Chromis chromis* and *Atherina* sp., were more abundant in NR relative to AS (Table 3.3.).
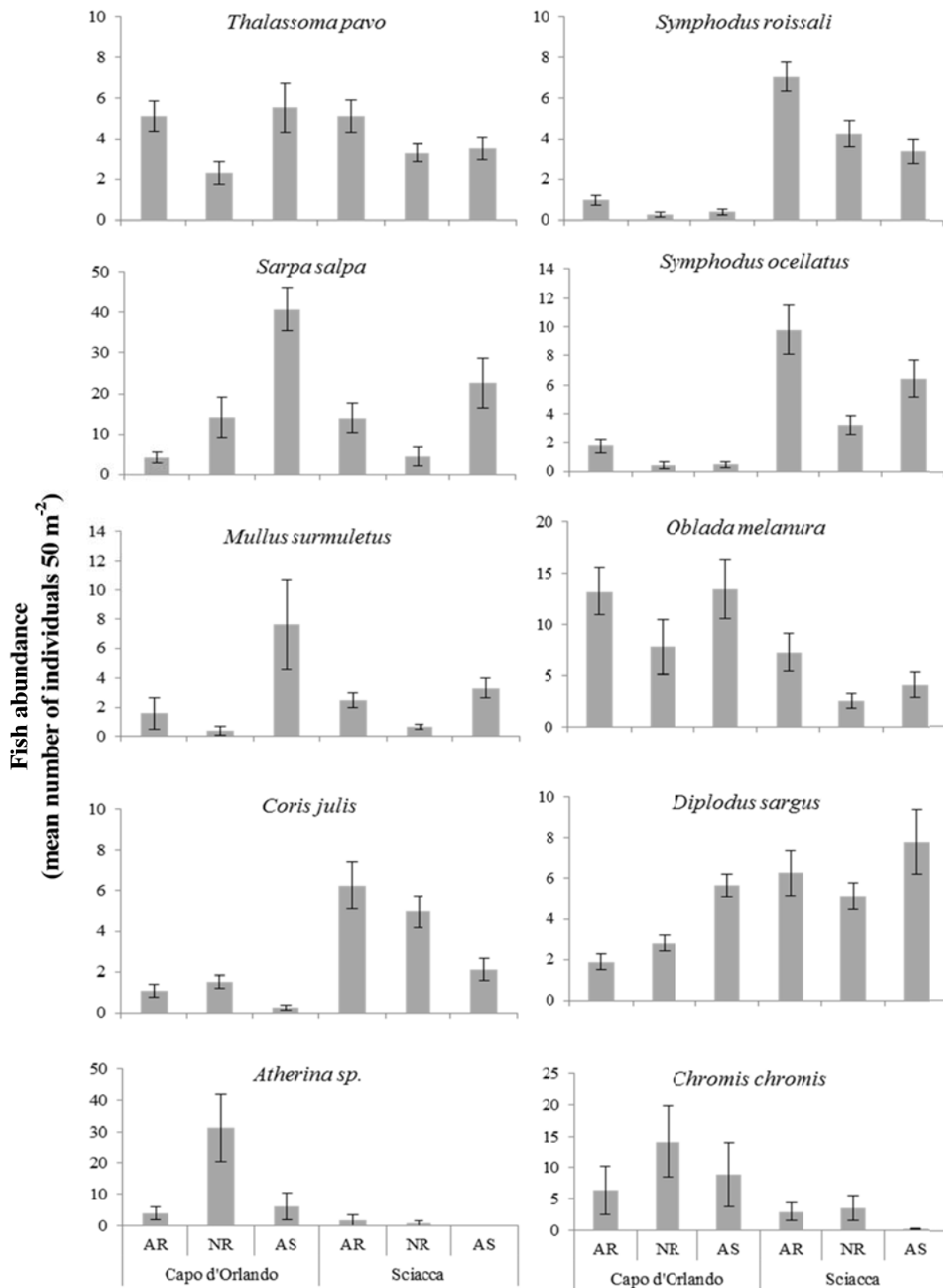
Fig. 3.3. Mean abundance (± S.E.) of the most abundant fish taxa recorded at different habitat and locations (abbreviations as in Table 3.1.).
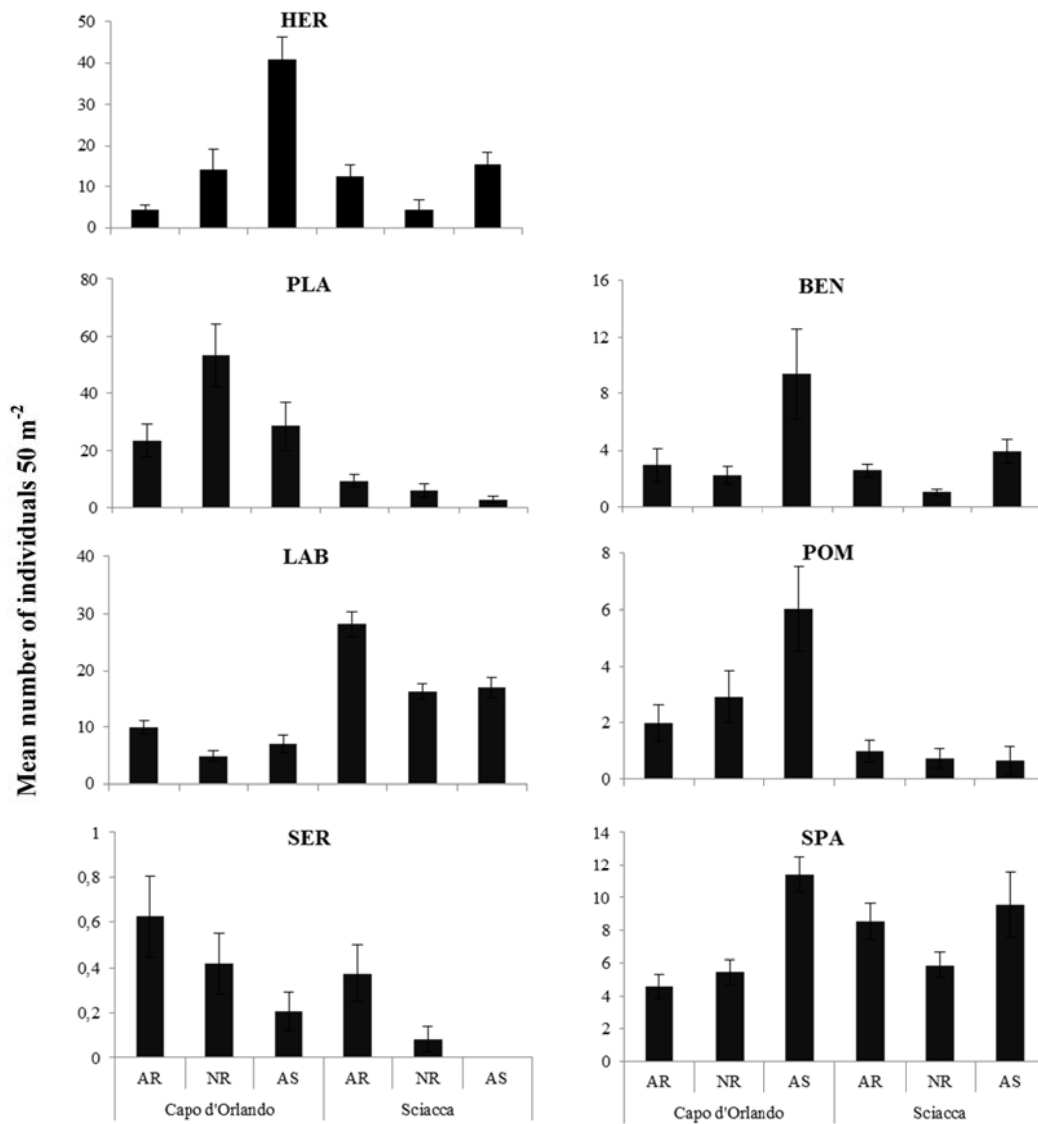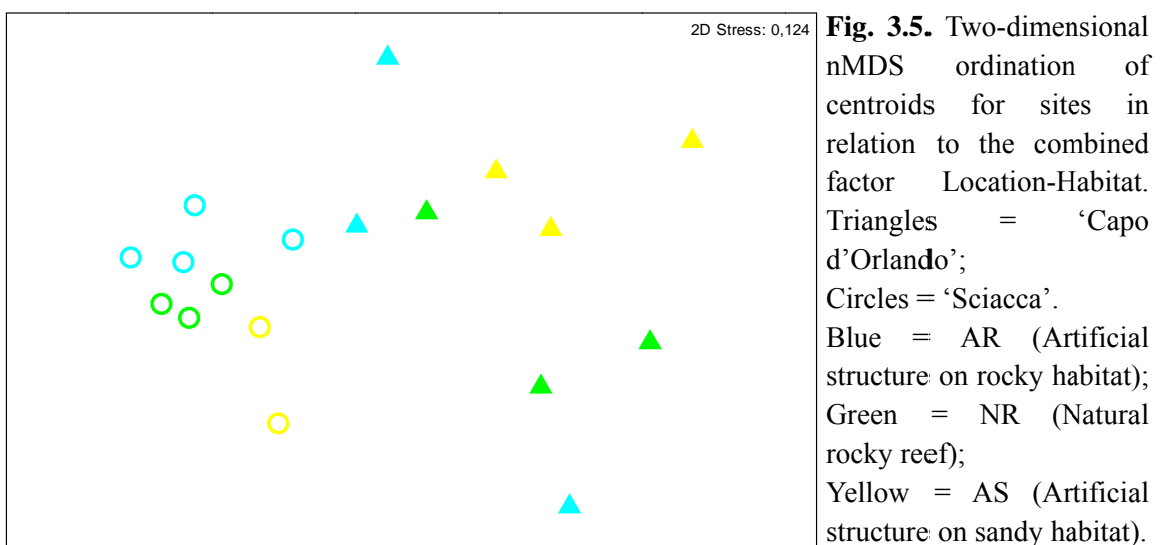
Fig. 3.4. Mean abundance (± S.E.) of the ecological categories (see materials and methods) of fish censused at different habitat and locations (abbreviations as in Table 3.1.).

Fig. 3.5. Two-dimensional nMDS ordination of centroids for sites in relation to the combined factor Location-Habitat. Triangles = 'Capo d'Orlando'; Circles = 'Sciacca'. Blue = AR (Artificial structure on rocky habitat); Green = NR (Natural rocky reef); Yellow = AS (Artificial structure on sandy habitat).

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Ecological categories, species richness and total abundance

The univariate analysis performed on the total abundance for each ecological category found significant differences for the factor habitat in BEN (*pseudo*-F = 15,26; p = 0,02), LAB (*pseudo*-F = 116,58; p = 0,008) and SER (*pseudo*-F = 5,7096; p = 0,035) (Table 3.4.). Pairwise test for the factor habitat revealed different trends for each of those latter EC: BEN differs among AS and NR (t = 16,292; p = 0,003), LAB differs among AR and NR (t = 25,045; p = 0,026), whereas SER lightly differs among AR and AS (t = 7,6404; p = 0,048).

In contrast, there were no detectable differences in species richness among habitats (*pseudo*-F = 2,9817; p = 0,257) on the overall fish assemblage. The analysis on the total abundance didn't detect any general trend (Habitat: *pseudo*-F = 0,6463; p = 0,681), but there were significant differences among habitats in each location separately (Habitat x Location *pseudo*-F = 5,693 p = 0,0139). Pair-wise test revealed that in location 'CDO', the total abundance was different in the comparison AR *versus* AS, whereas in the location 'SCI' it differed between NR *versus* AS (Table 3.4.).

Table 3.2. Results of the Permutational multivariate analysis of variance for the structure of the fish assemblage based on the Bray-Curtis dissimilarity measure for log(x + 1) transformed abundance data. The test was done using 9999 permutations of residuals under a reduced model. LO = Location; HA = Habitat; SI = Site; other abbreviations as in Table 3.1. * = P < 0,05; ** = P < 0,01, ***=P<0,001, ns= not significant.

	df	MS	Pseudo-F	Pairwise Comparisons	
LO	1	42253	10,452***	AS *vs* NR	*
HA	2	9196,3	3,2803*	AR *vs* AS	ns
LOxHA	2	2769,9	0,68516	AR vs NR	ns
SI(HAxLO)	12	4042,8	4,0514***		
Res	126	997,87			
Total	143				

Table 3.3. Results of SIMPER analysis showing fish species contributing most (percentage contribution > 5%) to dissimilarity (Diss) between habitats (abbreviations as in Table 3.1.).

Species	Mean abundance		Diss/SD	% contribution to diss
	NR	**AS**		
Average diss = 53,56				
Sarpa salpa	1,29	3,16	1,49	13,94
Oblada melanura	1,14	1,74	1,25	9,01
Chromis chromis	0,97	0,58	0,83	7,68
Mullus surmuletus	0,27	1,08	0,92	6,78
Aterina sp.	0,86	0,34	0,61	6,66
Thalassoma pavo	1,08	1,34	1,14	6,57
Mugil sp.	0,52	0,95	0,95	6,25
Diplodus sargus	1,41	1,8	1,2	5,69
Coris julis	1,13	0,44	1,16	5,43
Diplodus vulgaris	0,72	1,22	1,09	5,42
	AR	**AS**		
Average diss = 51,13				
Sarpa salpa	1,44	3,16	1,47	14,32
Oblada melanura	1,76	1,74	1,21	8,62
Diplodus sargus	1,29	1,8	1,27	7,26
Mullus surmuletus	0,74	1,08	0,93	6,61
Thalassoma pavo	1,62	1,34	1,29	6,26
Chromis chromis	0,7	0,58	0,78	6,11
Mugil sp.	0,52	0,95	0,88	5,97
Symphodus ocellatus	1,44	0,77	1,07	5,89
Diplodus vulgaris	1,02	1,22	1,21	5,81
Coris julis	1,18	0,44	1,05	4,91
	AR	**NR**		
Average diss = 52,24				
Sarpa salpa	1,44	1,29	1,16	11,39
Oblada melanura	1,76	1,14	1,2	10,2
Chromis chromis	0,7	0,97	0,83	8,49
Symphodus ocellatus	1,44	0,7	1,19	6,88
Thalassoma pavo	1,62	1,08	1,12	6,87
Aterina sp.	0,28	0,86	0,51	6,35
Diplodus sargus	1,29	1,41	1,16	6,29
Diplodus vulgaris	1,02	0,72	1,25	5,96
Coris julis	1,18	1,13	1,22	5,84
Mugil sp.	0,52	0,52	0,78	5,15

	df	MS	Pseudo-F	MS	Pseudo-F	MS	Pseudo-F
		TOTAL ABUNDANCE		**SPECIES RICHNESS**		**PLA**	
LO	1	39,295	3,8099	25,01	0,86969	333.05.00	21.511**
HA	2	28,614	0,38711	24,059	3,0008	10.463	0.40751
LOxHA	2	73,918	7,1668*	8,0174	0,27879	25.94	16.731
SI(HAxLO)	12	10,314	2,5254**	28,758	9,3449***	15.504	2.422**
Res	126	4,0841		3,0774		64.013	
Total	143						
		LAB		**BEN**		**SER**	
LO	1	177.65	21.767**	22.403	0.28769	12.952	41.755
HA	2	21.187	116.58**	13.202	14.767*	11.404	31.784*
LOxHA	2	0.17934	0.021975	0.88616	0,790277778	0.035321	0.11387
SI(HAxLO)	12	81.612	10.728***	7.787	7.7758***	0.31019	14.677
Res	126	0.76077		10.014		0.21135	
Total	143						
		SPA		**HER**		**POM**	
LO	1	0.87889	0.22347	36.321	26.044	29.54.00	6.0644*
HA	2	10.727	20.617	124.22.00	28.185	38.917	0.78698
LOxHA	2	51.325	1.305	43.237	31.003	49.717	10.207
SI(HAxLO)	12	39.329	5.0744***	13.946	3.7849***	4.871	5.2345***
Res	126	0.77505		36.846		0.93055	
Total	143						

Discussion

Results from multivariate analyses identified significant differences in the composition of the fish assemblage among the three habitats considered. Interestingly fish assemblages did not differ between artificial substrata in a rocky context and nearby natural reef, while this was true for fish assemblages associated to artificial habitats in sandy environments.

In Italy, most of the studies regarding the ecology and distribution of fish assemblage inhabiting artificial substrates in shallow coastal waters were carried out on sandy coastlines lacking natural rocky reference conditions (Guidetti, 2004; Cenci et al., 2011). However, a study conducted by Clynick (2006) in the north-east Tyrrhenian sea, compared fish assemblages among marina's breakwaters and nearby natural reef founding no

differences. Those results confirmed previous studies, where reported a similarity in the fish assemblages among artificial and natural reefs (Ambrose & Swarbrick, 1989 and reference therein). The fundamental implication of artificial structures that support similar assemblages to those inhabiting adjacent rocky shore would be a reduction of their impact, i.e. loss or fragmentation of natural habitats (Bulleri, 2005).

On the other hand, resulted that artificial structures deployed on a sandy bottom host fish assemblage differing in structure from those inhabiting natural rocky shores. Burt et al. (2012) recently conducted a study on the fish community structures comparing breakwaters and natural reef, in a tropical area. Although not explicitly specified, the breakwater under exam was in a prevalent sandy bottom. Consistently with the results of this study, they found that breakwater habitat hosted fish communities clearly distinct from that of the natural reef.

The differences found here are mainly driven by sedentary rather than mobile species, such as the benthic taxa that showed clear differences in abundance among artificial structures in a sandy habitat and adjacent natural reef. Within those benthic taxa, the blenny species and *Mullus surmuletus* were the most abundant taxa. Low abundance or even absence of other families of benthic fishes (Gobiesocidae, Scorpaenidae and Tripterygiidae) associated to breakwaters was also reported by Santin & Willis (2007) for the north Adriatic sea. This result is common for several studies carried out in Mediterranean rocky reefs and breakwaters where the Blenniidae resulted the most diverse and abundant group among the cryptic fishes (Illich & Kotrschal, 1990; Macpherson, 1994; Lipej & Richter, 1999; La Mesa et al., 2004). Those sedentary benthic and cryptic species, aggregate in those artificial environment where they found crevices for shelter and resources (Santin & Willis, 2007; Pérez-Ruzafa et al., 2006). In particular the blennies species are characterized by very low mobility (Harmelin, 1987) and are likely unable to cross the hundreds of meters of wide sandy bottom to the next hard surface. Indeed the artificial structures could act as fish attractors, hosting individuals previously inhabiting other hard substrata or alternatively they act as fish producers, enhancing fish biomass in an area by providing new spaces, refuges and habitats (Carr & Hixon, 1997; Cenci et al., 2011).

Furthermore, the clear separation among fish assemblages inhabiting the two locations was not unexpected considering that they were deployed in separate seas (strait of Sicily and southern Tyrrhenian sea) with different oceanographic characteristics (Sanfilippo et al., 2011). In effect, at very large spatial scales (i.e. hundreds to thousands of km), substantial differences in abundance of adult reef fish assemblages are expected (García-Charton et

al., 2000; García-Charton et al., 2004; La Mesa et al., 2011). This regional variation in fish assemblage can be due to several, non-exclusive causes, such as large-scale climatic differences, habitat structures - resulting in differential availability of resources such as food or shelter, larval dynamics and recruitment variability (García-Charton & Pérez-Ruzafa, 1999; García-Charton et al., 2000; García-Charton et al., 2004). But even higher differences have been found at the lowest spatial scale, i.e. sites (hundreds to thousands of meters), as typical of the variability in the Mediterranean fish populations (García-Charton et al., 2000).

In this study the species richness and total abundance among the three habitat types was similar at both sampling locations. Some authors stated that species richness and abundance may be related to connectivity (Airoldi et al., 2005; Bulleri & Airoldi, 2005; Cenci et al., 2011), since higher fish abundances and richness have been found on submerged artificial reefs with higher connectivity rather than on more isolated ones (Vega Fernández et al., 2008).

In conclusion, the ecological implications for biodiversity of fish assemblages (at local and regional scales) depend on the type of natural habitat mostly affected. Specifically, they will vary between the case in which artificial structures are deployed on hard-bottoms or soft-bottoms.

Artificial structures deployed on soft-bottom function as an island that aggregate rocky fishes but with a limited connectivity with other artificial structures and rocky shores, developing assemblage of fish with peculiar characteristic in terms of structure and composition. When coastal defences are built in a rocky setting they might function as stepping stones supporting connectivity.

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CHAPTER 4

Effects of detrital enrichment from artificial defence structures on the adjacent soft–sediment macrofauna

Introduction

Artificial defence structures have been widely used worldwide for protecting coasts from erosions and inundations. The greater threats given by climate change, storm surges, and sea level rise may lead to an increase of artificial coastal-defences (i.e. breakwaters and seawalls) with consequent impacts to coastal habitat and communities (Dugan et al., 2011). Marine artificial structures may cause profound changes to the native coastal ecosystems. This is particularly relevant for coastal infrastructures built on prevailingly sandy bottoms (Dugan et. al, 2011), where they cause the direct fragmentation and loss of the habitats through the conversion to artificial hard substrata. They also alter the hydrodynamic and depositional environment, therefore introducing uncharacteristic changes in adjacent soft bottom habitats (i.e. grain size, content of organic matter, redox conditions, and associated native assemblages of animals and plants; Davis et al., 1982; Barros et al., 2001; Airoldi et al. 2005; Martin et al.; 2005, Bertasi et al., 2007; Colosio et al., 2007). These direct effects are most evident just around the infrastructures, on the seaward-sheltered sides related to the increased wave energy and on the landward-exposed sides due to decreased wave energy (Santin & Willis, 2007 and references therein).

The introduction of hard coastal structures can also lead to indirect changes to sedimentary environments related to the widespread introduction of hard-bottom species into areas where they are naturally scarce. Such an introduction could have important indirect consequences for the functioning of these dynamic environments, through the unnatural subsidy of detrital material that is sloughed off the artificial structure. Moreover artificial structures could entrap drifting algae and other detritus, further contributing to organic build up in the sediments. There is growing recognition that the export of detritus to adjacent habitats is an important form of connectivity among coastal systems that can influence local and regional productivity and the spatial organization of marine ecosystems (Polis et al., 1997, Loreau et al., 2003, Marczak et al., 2007; Krumhansl & Scheibling, 2012).

In subtidal soft-sediment habitats detritus represent a primary source of food, attracting diverse assemblages of detritivores, predators, microbes, then influencing their distribution and secondary production (Krumhansl & Scheibling, 2012 and reference therein). These effects depend on the availability, size and resident time of the deposits (Norkko et al., 2000), but could also vary depending on which benthic components (e.g. animal vs vegetal) will prevail on the infrastructures.

Previous work has explored the ecological factors that can lead to the prevalence of each of these two components in the system, identifying the complex interactions between these two dominant groups (Bulleri et al., 2006) and revealing that opportunistic macroalgae tend to be favoured by the severe human disturbances typical of these environments (Airoldi et al., 2005; Bulleri & Airoldi, 2005; Airoldi & Bulleri, 2011).

I have analysed possible changes in sedimentary environments and associated assemblages related to the unnatural introduction of detritus from hard-bottom species colonising artificial defence structures along the sedimentary coastline of the north Adriatic sea (Italy). In this region, which is naturally devoid of rocky substrata, more than 190 km of rock armoured infrastructures (mainly breakwaters, groynes, seawalls and jetties) have been built in the past 40 years along about 350 km of coastline (Bondesan et al. 1995), introducing about 2 km^2 of artificial hard substrata along these shallow, moderately exposed sandy bottoms.

Information on the dominant epibenthos species inhabiting the artificial substrate of the defence structures in the study area was available from Bacchiocchi & Airoldi (2003) and Santin & Willis (2007). However it was essential for this study to identify the type and quantify the amount of detritus originated by the dominant benthos associated to the structures, and describe its spatial distribution on sedimentary environments surrounding the defence structures. A field manipulative experiment was subsequently carried out in order to assess the different effects of detritus from *Ulva* spp. and *Mytilus galloprovincialis* (the two main producers of detritus on the structures) on soft-bottom associated assemblages. I predicted different effects of these two forms of detritus because of the nature of the organic material (specific gravity and chemical content) and the provision of shell material associated with mussel deposition. Specifically, I analysed whether effects related to wrack accumulation differ depending on the species producing the detritus. I considered both the responses of the macrofauna inhabiting the sediment as well as possible effects related to the attraction of detritus feeding species that are directly

associated to the detritus; I also tested if any observed effects were consistent over time, by repeatedly sampling 3 times over a short temporal scale.

Materials & Methods

Study Area

The surveys and the manipulative experiment were carried out in June 2009 on the artificial defence structures located at Lido di Dante (44°23'10''N, 12°19'10''E), Adriatic Sea (Fig. 4.1.). The North-Western Adriatic coastline is a sandy flat system, protected for more than 190 km by numerous defence structures as a measure against erosion. The beach of Lido di Dante (Fig. 4.2.) is located 12 km south of the port of Ravenna (Italy) between the mouths of the Fiumi Uniti (0,9 km North) and Bevano rivers (2,6 km South), with a tidal amplitude ranging between 0,30 m and 0,85 m, a gentle seabed slope of about 6 m/km, and bottom sediments ranging from well-sorted fine to medium sand (Bertasi et al., 2007).

The artificial defence system of Lido di Dante is comprised of groynes and offshore detached breakwaters, built with large blocks of limestone, about 0,5–2 m in diameter. In particular, the groynes consisted of about 70-m-long structures, perpendicular to the shore, that were built in the early 1980s. The breakwaters consisted of two low-crested structures (i.e. crest emerging at low tides) parallel to the shore, about 350 m long and about 150 m from the shoreline, that were built in 1996 (Bertasi et al., 2007).
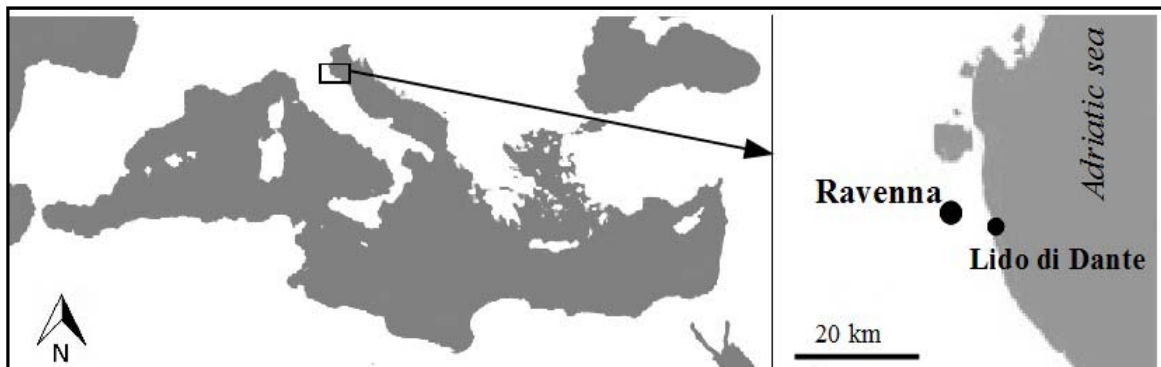

Fig. 4.1. Map of the study area where the surveys and the field experiment were carried out.

A detailed description of the hard-bottom benthos associated with breakwaters and groynes along the Ravenna's shoreline, is reported in Bacchiocchi & Airoldi (2003). Assemblages are relatively species poor, with a strong dominance by few taxa. Animals included *Mytilus galloprovincialis*, *Ostrea edulis* and *Crassostrea gigas*, Serpulidae, *Chthamalus stellatus*, Balanidae, Ascidiiae and Actinidae. Algae included *Ulva intestinalis*, *Ulva laetevierens* and filamentous species (among which *Cladophora vagabunda* and *Polysiphonia breviarticulata*). Encrusting algae are usually rare and included *Ralfsia verrucosa*. Bacchiocchi & Airoldi (2003) reported average coverage by the two dominant groups, *Mytilus galloprovincialis* and *Ulva* spp., of 74 %, and 29 %, respectively. Moreover patches of bare rock are reported to be frequent in the assemblage, comprising on average about 4 % of the substratum, but peaking up to 35% at some times of the year.

Fig. 4.2. Aerial view of the shoreline and the coastal-defence system at Lido di Dante (Ravenna, Italy). Photo by Giorgio Benelli, published with permission.

Surveys of the nature, amount and distribution of detritus around structures

The surveys were planned to analyse the composition, the amount and the distribution of detritus produced by the dominant benthic species colonizing the artificial defence structures at Lido di Dante.

The first survey was performed in order to describe the distribution of the different detritus types over the surface of the sediment adjacent the breakwaters. The description was done by scuba diving along 50-m-long transects deployed on the bottom sediments, adjacent and parallel to the breakwater. Two transects were sampled at both the landward and seaward sides of the breakwaters. The distribution of each of 8 detritus categories ("Dense Mytilus"; "*Ulva*; Mytilus & Sand"; "Mytilus & Shells"; "Shells & Sand"; "Anemonies, *Mytilus* & Sand"; "Oysters & *Mytilus*"; "Sand"; "Rubble") was quantified by recording the distance at which changes in each detritus category occurred along the transect. The distance data were subsequently converted into average percentage cover for each category, by dividing the total amount of metres covered by each detritus type for the total length of the transect.

An additional survey was conducted in order to evaluate the extension of the detritus mat around the artificial defence structure. The survey was carried out by scuba diving along both the landward and seaward sides of two breakwaters. Six 50-m-long transects were deployed at the bottom, adjacent and perpendicular to each of the two sides of the breakwaters. Each transect was 10 m apart. The extension of the detritus mat was quantified by recording the maximum distance from the breakwater at which patches of detritus were observed.

Finally, the mass of detritus per unit surface was estimated by using sediment corers with a diameter of 13 cm and a height of 15 cm, that were pushed approximately halfway into the sediment. Six replicated corers were collected at each of six patches of either mussel or *Ulva* detritus, selected at random along the transects that was deployed adjacent and parallel to the breakwater.

Experimental procedure

The experiment was set-up in June 2009 because at this time of the year detritus tends to accumulate more persistently in shallow habitats due to periods of calm sea conditions (Vetter, 1996; Okey, 2003). Specimens of *Mytilus galloprovincialis* (hereafter *Mytilus*) and *Ulva* spp. (mainly *Ulva latevierens* and *Ulvaa intestinalis*; hereafter *Ulva*) were collected alive from the artificial defence structure in Lido di Dante and transported to the laboratory

for the preparation of the nets to be used in the experiment. Once in the laboratory the *Mytilus* was frozen for 24 hours, in order to mimic the natural death, which occurs following the detachment from the structure and smothering in the sediments. Conversely, *Ulva* was stored at 4°C for a maximum of 24 hours, to preserve it fresh until the deployment of the experiment in the field. The detritus from *Mytilus* and *Ulva* were disposed into nylon mesh bags (60 x 60 cm, 1 cm mesh size) to be used for the manipulative experiment. The experiment comprised three treatments: '*Ulva*' detritus (0.5 kg wet weight per bag), '*Mytilus*' detritus (3 kg wet weight per bag) and un-manipulated Controls (Fig. 4.3.).
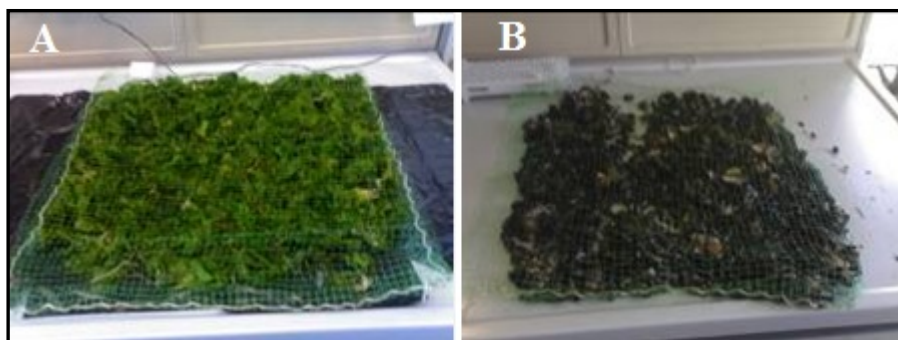
Fig. 4.3. Sample of the net bags containing A) *Ulva* and B) *Mytilus* detritus used in the experiment.

The amount of Mussel detritus in each treatment was chosen based on the measures done during the field survey (see Results). The amount of *Ulva* measured in the survey (see Results) was however much smaller than what occurring generally at time of the year (see Fig. 4.4.) on the breakwaters. For this reason the amount of detritus from *Ulva* used in the experiment was increased by 5 times compared to the values measured in the present study.

The experiment was deployed in an area 50 m apart from the artificial defence structure. The area was located at 4 m in depth and at 200 m off-shore, which are consistent with the real conditions of the soft bottom adjacent to the artificial defence structure. For each of the three treatments 12 plots were established, for a total of 36 plots randomly distributed. The sampling was carried out at three times (specifically, T1 at 2 days after the deployment of the net bags, T2 after 7 days and T3 after 21 days) to test whether the responses to treatments were persistent over time. At each sampling time, 4 plots for each treatment were randomly selected and sampled, except for *Ulva* at T3 that accounted for plots. In fact one plot of *Ulva* was lost at this time, probably because removed by waves.

Variables measured in each plot included the soft-bottom macrofauna (here after SB-macrofauna) and the macrofauna directly associated to the detritus (here after DE-macrofauna).

The SB-macrofauna was sampled by collecting 4 cores (10 cm diameter, 10 cm deep) of sediment below the net-bags. In the laboratory, each core was washed in a 500 μm sieve, and the material retained was preserved in a 7% formalin solution. The material from each sieved core was then carefully sorted under a stereomicroscope and identified to the lowest possible taxonomic level.

The DE-macrofauna was sampled by recovering the net-bags and keeping them into hermetic plastic packets. In the laboratory, the macro-invertebrates were extracted from bags and detritus, and then preserved, sorted and identified as described previously.

Furthermore, the mass loss of detritus at each sampling time was evaluated as remaining wet weight for both *Ulva* and *Mytilus.*

Data analyses

The weight loss of *Ulva* detritus across the experiment was calculated as remaining wet weight (g) of the initial mass (500 g) at each sampling times. For the *Mytilus* detritus, the organic matter (OM) contained in samples of 200 g (wet weight) of dead *Mytilus* was determined before the start of the experiment (T_0) and at the following sampling times. The samples of *Mytilus* were placed in ceramic cups, dried in an oven for 24 h (100°C), weighed (Dry Weight, DW), burned to ash in a furnace for 6h (500°C) and re-weighed (Ash Dry Weight, ADW). The Ash-Free Dry Weight (AFDW) was calculated as the difference ADW-DW. The organic matter content was calculated as (DW/AFDW)*100, and expressed as the percentage of the oven dry weight (DW).

The effects of detritus enrichment from *Mytilus* vs *Ulva* on soft-bottom macrofauna were tested using both univariate and multivariate analyses. The mixed model used in these analyses included the factors 'treatment' (fixed, 3 levels: '*Ulva*', '*Mytilus*', 'Control') and 'time' (random, 3 levels: T1, T2, T3). The effects of different detritus types for detritus feeding macrofauna found associated to the detritus were tested following the same rationale but including only two 'treatment' levels: ('*Ulva*', '*Mytilus*'). Each treatment comprised 4 replicates (plots) for each time except for one *Ulva* treatment at time 3 where only 3 replicates were available.

The effect of the detritus enrichment at different sampling times on the assemblage structure of SB- and DE-macrofauna was tested by Permutational analysis of variance

(PERMANOVA) (Anderson, 2001) according to the designs described above. Multivariate PERMANOVA used Bray-Curtis similarity matrices of square root transformed abundance data with 9999 permutations of residuals under a reduced model (Anderson, 2001; Clarke & Warwick, 2001). For some terms in the analysis, there were not enough permutable units to get a reliable test by permutation, so a p-value was obtained using a Monte Carlo test (Anderson & Robinson, 2003). Furthermore, the nMDS ordinations (non-metric multidimensional scaling) were produced to visualize possible patterns among treatments.

Differences in species richness and the total abundance of macroinvertebrates between treatments were analysed by permutational ANOVA (using the statistical package PERMANOVA+ for PRIMER, Anderson et al., 2008). The analyses were run on a matrix of Euclidean distances calculated from the original raw data, and P-values were estimated using 9999 random permutations of the appropriate exchangeable units.

When multivariate analyses detected significant differences among factors, the procedure SIMPER (Clarke & Warwick, 2001) was used to identify the taxa which most contributed to the percentage dissimilarities among assemblages. Only species that contributed more than 5% to the average dissimilarity between treatments were considered as good discriminating species.

PERMANOVA was run using the type III of Sum of Squares (Anderson, 2001). All analyses (PERMANOVA, MDS and SIMPER) were performed using PRIMER with PERMANOVA+ (v. 6, PRIMER-E Ltd., Plymouth, UK).
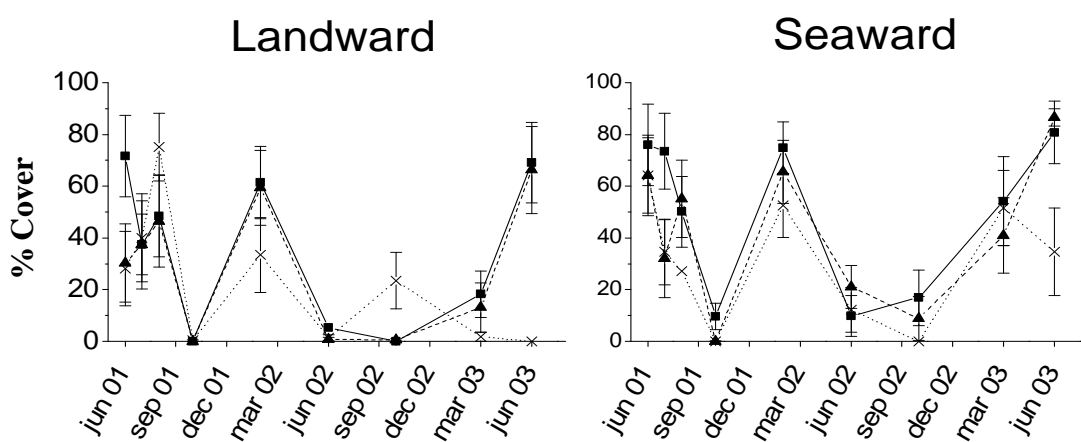

Fig. 4.4. Mean percentage cover (± S.E. n=8) of Ulvales at Lido di Dante on landward and seaward sides of the coastal-defence structure as a function of time, at different seasons through the years 2001, 2002, 2003 (F. Bacchiocchi, PhD Thesis, 2003).

Results

Survey

The composition of the detritus mat is visualized in the fig. 4.5.. In the landward side of the breakwaters "Mixed Mytilus & Shells" are the main group with a coverage of 33, 5%. Follow "Shells & Sand" and "Dense *Mytilus*" with a coverage respectively of 23,5 % and 22 %. Finally "Sand and *Mytilus*" and "Sand" cover the 11, 3 and 9,7 % respectively. At the seaward side of the breakwater "Mixed *Mytilus* & Shells" comprise 34,1 % of the detritus coverage, followed by the categories "Shells & Sand" and "Sand" with a coverage of 31,3 % and 22,8 % respectively. Then "*Ulva, Mytilus* and Sand" with 5 % coverage. Finally "Anemonies, *Mytilus* & Sand" and "Mytilus & Sand" account respectively for the 4 % and 2,8 % of the coverage.
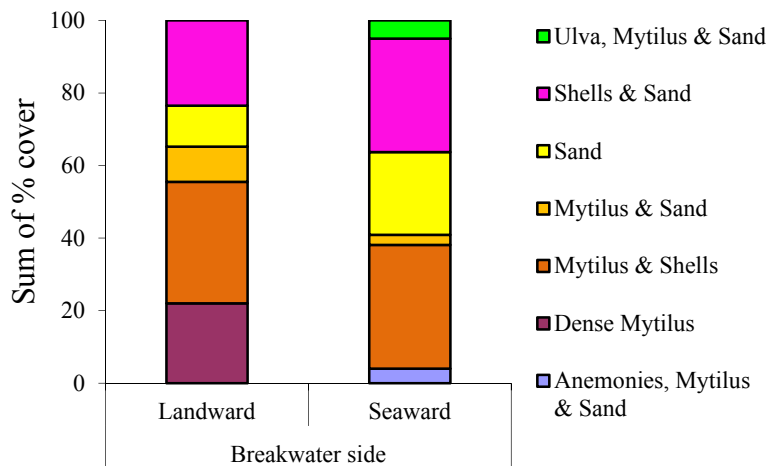
Fig. 4.5. Distribution of the qualitative features that compose the detritus mat around the breakwaters at Lido di Dante. The survey included both the landward and the seaward side.

The detritus mat extended on average 5,4 m ± 1,3 S.E. far from the breakwater at the landward side and 3,2 m ± 0,7 S.E. at the seaward side.

The detritus comprised on average a wet weight of 7,5 kg/m^2 ± 3,6 S.E. of mussel shells at the landward side and 6,7 kg/m^2 ± 3,1 S.E at the seaward side of the breakwater, which corresponds approximated to 3 kg for 3600 cm^2 (the surface of the experimental plot). Accumulation of detritus from *Ulva* was observed only in the very sheltered conditions offered by the presence of groyns in Lido di Dante. Here *Ulva* presented on average a weight of 0,3 kg/m^2 ± 0,1 S.E., which corresponds to ~ 0,1 kg for 3600 cm^2. However our previous observations in the region over many years suggested that 2009 was characterized by unusually low coverage of *Ulva* relative to the average (Fig. 4.4.). Therefore I increased the amount of *Ulva* in the treatments at 0,5 kg to simulate quantities of detritus more typical for the study region.

Species richness and total abundance

The total number of taxa and the total number of organisms were not affected by the detritus enrichment both for SB-macrofauna (species richness: *pseudo*-F = 0,887; p = 0,50; total abundance: *pseudo*-F = 0,183; p = 0,85) and DE-macrofauna (species richness: *pseudo*-F = 13,096; p = 0,13; total abundance: *pseudo*-F = 2,748; p = 0,30).

Detritus decomposition was very fast for both *Ulva* (Fig. 4.6) and *Mytilus* (Fig. 4.7). Two days after the start of the experiment (T_0) the remaining wet weight (W.W.) of *Ulva* was 26 % of the initial wet mass (T1; W.W. = 131,83 g ± 4,63 S.E.). *Ulva* weight loss continued dramatically during the following days, and only 6 % of the initial wet weight remained after 7 days (T2; W.W. = 28,95 g ± 14,23 S.E.). *Ulva* detritus was almost completely degraded after 21 days, where only 0,6 % of the initial wet weight remained on average (T3; W.W. = 3,9 g ± 2,2).

Concerning the detritus from *Mytilus*, the O.M. contained in 200 g of dead specimens at the start of the experiment (T_0) was estimated to be, as average, the 17,58 % ± 0,52 S.E. of the dry weight. After 2 days (T1) of experiment exposition the organic matter became the 6,89 % ± 0,12 S.E. of the dry weight, that correspond to the 39 % of the initial O.M.. The decrease of the mussel's organic matter continued during the first week (T2; O.M. = 4,03 ± 0,21 S.E.), when organic content was reduced to 23 % of the initial O.M.. After 21 days the organic matter was reduced to 21 % of the initial O.M. (T3; O.M. = 3,8 % ± 0,06 S.E.).
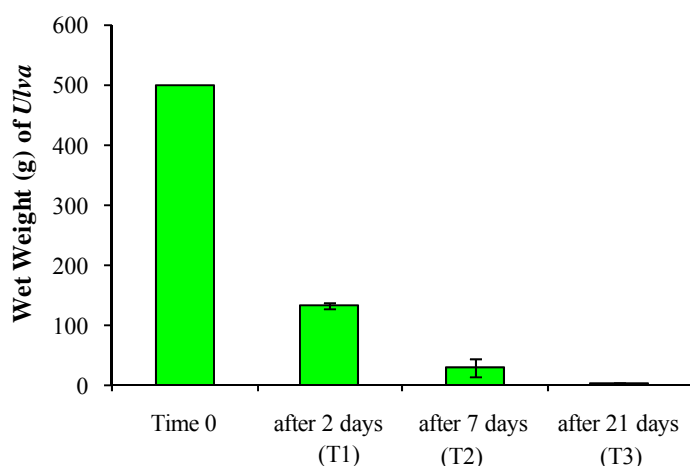

Fig. 4.6. Wet weight (g) of *Ulva* contained in the experimental mesh bag at the start of the experiment (T_0) and compared with the average ± S.E. of the wet weight of *Ulva* remaining in the mesh bags in the following sampling times.
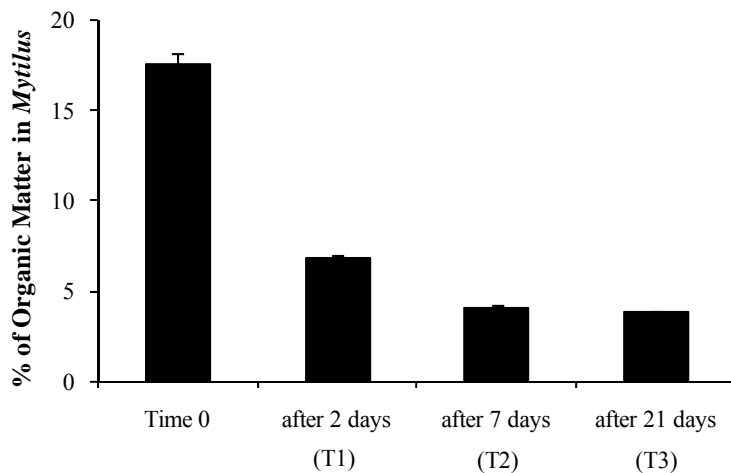
Fig. 4.7. The organic matter content in *Mytilus*, at the beginning of the experiment (T_0) and in the following sampling times, was calculated as the proportion of AFDW to DW and converted to percentage.

Macrofauna associated to the soft-sediment

Concerning the SB-macrofauna, a total of 5254 individuals belonging to 42 families was recorded. Among them *Lentidium mediterraneum* (family Corbulidae) resulted the dominant species, with an average of 80 individuals per sample.
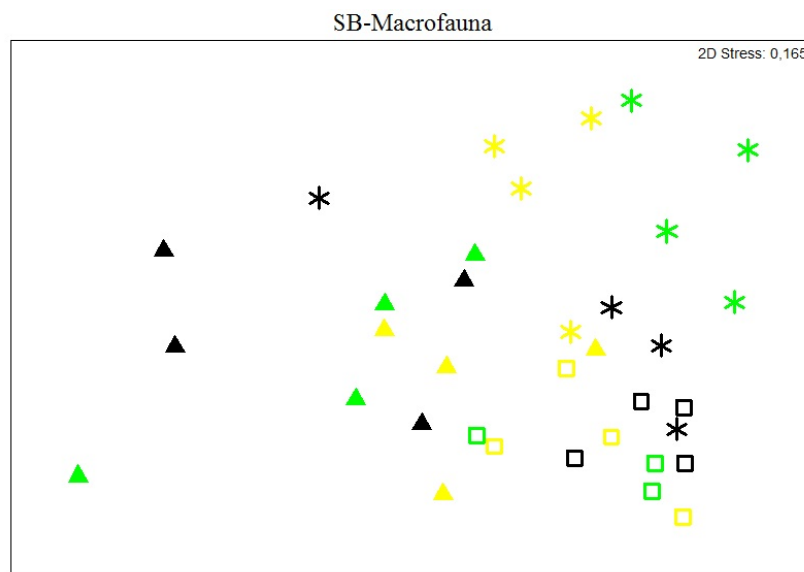

Fig. 4.8. SB-Macrofauna. Two-dimensional nMDS ordination for for the combined factor Time-Treatment. Green: *Ulva*, Black: *Mytilus*; Yellow: Control; Triangles: T1; Stars: T2; Quadrates: T3.

The experimental manipulation of detritus initially affected the structure of macro-invertebrate assemblages, but the effects were not persistent over time (time × treatment interaction; *pseudo*-F = 1.6, p = 0.019; Table 4.1.). The nMDS showed that macrofauna differed between treatments at T1 and T2 while group differences were no longer noticeable at T3 (Fig. 4.8.). However the post hoc pair-wise test was not enough powerful to reveal significant alternatives to the null hypothesis (Table 4.1.).

SIMPER (Table 4.2.) revealed that *Lentidium mediterraneum,* Veneridae and Capitellidae were far less abundant in both *Ulva* and *Mytilus* detritus treatments compared to Control plots at time T1. A comparison between *Mytilus* and *Ulva* showed that *Lentidium mediterraneaum* and Capitellidae presented an higher abundance in *Mytilus*, whereas Veneridae in *Ulva.*

After 7 days *Lentidium mediterraneum* became more abundant in both *Ulva* and *Mytilus* treatments compared to Control plots. *Mytilus* plots comprised a higher abundance of *L. mediterraneum* with respect to *Ulva* plots. At this time the amphipods (i.e. Isaeidae, Dexaminidae and Gammaridae) as well as the crabs belonging to the family Portunidae showed the highest abundance at the *Ulva* treatment with respect to both Control and *Mytilus*. Capitellidae showed an higher abundance at the Control with respect to both *Ulva* and *Mytilus.*Veneridae was more numerous in *Ulva*and *Mytilus* with respect to the Control. After 21 days the overall abundance of *Lentidium mediterraneum* reached 89.5 % of total abundance of the sampled organisms. *Lentidium* was slightly more numerous in detritus addition treatments, particularly the *Mytilus* ones, with respect to Control.

Table 4.1. Results of the PERMANOVA for the structure of the assemblage of macrofauna associated to the soft-bottom (SB-Macrofauna). The analysis is based on the Bray-Curtis dissimilarity measure for square root transformed abundance data. The test was done using 9999 permutations of residuals under a reduced model. In the cases of low permutations, P-values were obtained using the Monte Carlo test.

PERMANOVA SB-Macrofauna				
	df	MS	Pseudo-F	P(perm)
Time	2	6376	7,3747	0,0001
Treatment	2	1515	1,0533	0,4449
Time x Treatment	4	1440	1,665	0,0195
Res	26	864,6		
Total	34			

Pair-wise tests						
	Within level T1		Within level T2		Within level T3	
	t	P(MC)	t	P(MC)	t	P(MC)
Control vs *Mytilus*	1,4882	0,0972	1,0714	0,3489	0,9	0,4624
Control vs *Ulva*	1,7555	0,0563	1,5876	0,0632	0,7	0,7096
Mytilus vs *Ulva*	1,2933	0,1989	1,4268	0,1202	0,9	0,5234

Table 4.2. Summary of SIMPER analysis for macrofauna associated to the soft-bottom (SB-macrofauna) showing the species most contributing (percentage contribution > 5%) to the dissimilarities (diss) between treatments at different times.

Species	Mean Abundance			% contribution to diss		
	Control	*Mytilus*	*Ulva*	**Control vs *Mytilus***	**Control vs *Ulva***	***Mytilus* vs *Ulva***
AFTER 2 DAYS (T1)				**Av diss = 49,00**	**Av diss = 53,15**	**Av diss = 50,31**
L. mediterraneum	53,75	28,5	18	64,44	65,73	56,7
Veneridae	4,5	0,5	2,5	8,47	7,68	7,4
Capitellidae	5,75	4,75	0,5	10,78	9,16	14,6
AFTER 7 DAYS (T2)				**Av diss = 56,23**	**Av diss = 56,54**	**Av diss = 54,13**
L. mediterraneum	41,75	85,75	69,25	58,7	33,8	42,1
Isaeidae	10,75	1,25	31,5	9,6	19,1	21,4
Dexaminidae	9	3,75	18,5	8,7	13,7	11,1
Capitellidae	4,75	0,5	0,5	5,2	3,9	0,6
Veneridae	4	6,75	8,25	4,4	5,1	5
Portunidae	0,25	0,5	10	0,6	7,4	7,1
Gammaridae	1,25	0,75	9,25	1,9	8,6	8,6
AFTER 21 DAYS (T3)				**Av diss = 35,88**	**Av diss = 41,62**	**Av diss = 31,87**
L. mediterraneum	136,75	180,75	163,33	84,8	89	89,2

Macrofauna associated to the detritus

In the DE-macrofauna samples, a total of 49951 individuals belonging to 40 families was recorded. The assemblage was generally dominated by three families of Amphipoda. Specifically, the family Dexaminidae showed on average 1086 individuals per bag while the families Gammaridae and Isaeidae showed on average 176 individuals per bag.

The nMDS ordination (Fig. 4.9.) showed an aggregation of the assemblages associated to *Mytilus* and *Ulva* at T1 and T2. Whereas at T3 the assemblage structures seemed to be more dispersed suggesting a separation between the assemblages associated to *Mytilus* and those associated to *Ulva.*

The PERMANOVA test for differences in the macrofauna assemblages associated to *Ulva* versus *Mytilus* revealed significant differences across the sampling times (time × treatment interaction; *pseudo*-F = 2.4, p = 0.006; Table 4.3.).

The pair-wise test that revealed significant differences only at T3, confirming the pattern visualized by the nMDS plot (Fig. 4.9.)

The SIMPER (Table 4.4.) revealed that detritus experienced a rapid colonization by the macrofauna from the surrounding soft-bottom. Gammaridae, Dexaminidae and Isaeidae quickly colonized both *Mytilus* and *Ulva* just after 2 days.

After 7 days *Ulva* revealed on average higher values of abundances of amphipods (Gammaridae, Dexaminidae and Isaeidae).

After 21 days the amphipods Gammaridae and Isaeidae still showed the highest abundance in *Ulva*. The fossorial amphipods Haustoridae, scarcely found during the two previous sampling dates, became important in discriminating among treatments and showed the highest abundance in *Mytilus* treatment. *Lentidium* showed the highest abundance in *Mytilus* bags, where the polychaete Capitellidae was also abundant.
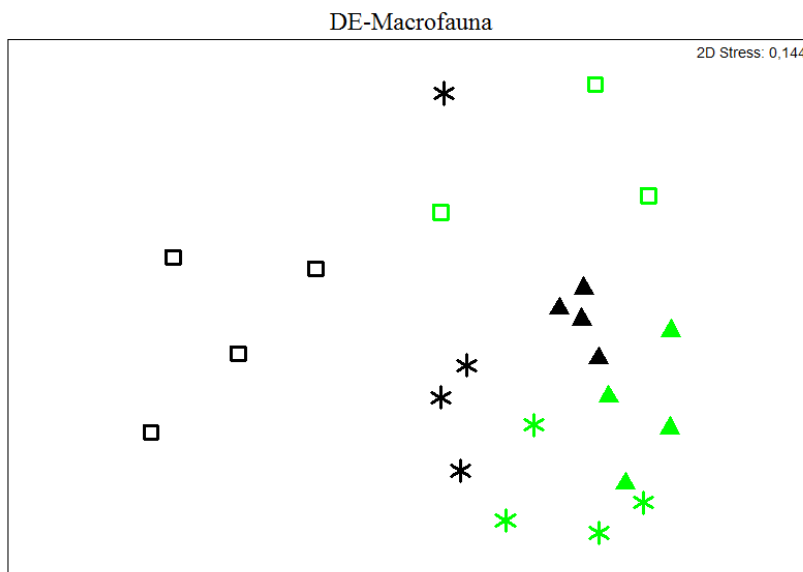

Fig. 4.9. DE-Macrofauna. Two-dimensional nMDS ordination for for the combined factor Time-Treatment. Green: *Ulva*, Black: *Mytilus*; Triangles: T1; Stars: T2; Quadrates: T3.

Table 4.3. Results of the PERMANOVA for the structure of the assemblage of macrofauna associated to the detritus (DE-Macrofauna). The analysis is based on the Bray-Curtis dissimilarity measure for square root transformed abundance data. The test was done using 9999 permutations of residuals under a reduced model. In the cases of low permutations, P-values were obtained using the Monte Carlo test.

	PERMANOVA DE-Macrofauna				
	df	MS	Pseudo-F	P(perm)	P(MC)
Time	2	8940	5,789	0,0001	
Treatment	1	6236	1,4709	0,285	
Time x Treatment	2	4246	2,7496	0,0043	
Res	16	1544			
Total	21				

	Pair-wise tests					
	Within level T1		Within level T2		Within level T3	
	t	P(MC)	t	P(MC)	t	P(MC)
Mytilus vs *Ulva*	1,3645	0,1771	1,6413	0,0814	2,0935	0,0222

Table 4.4. Results of SIMPER analysis for macrofauna colonazing (DE-macrofauna) the detritus showing the species most contributing (percentage contribution > 5%) to the dissimilarities (diss) between treatments at different times.

Species	Mean Abundance		% contribution to diss
	Mytilus	*Ulva*	
After 2 days (T1)			**Av diss = 53,52**
Dexamidae	230,25	2038,67	68,44
Isaeidae	265	226,67	20,09
Gammaridae	260	358,33	8,7
After 7 days (T2)			**Av diss = 67,99**
Dexamidae	759	4414	76,53
Isaeidae	204	412	8,11
Gammaridae	251	454	8,03
After 21 days (T3)			**Av diss = 81,24**
Isaeidae	47	280	33,3
Lentidium	139	0	22,97
Haustoridae	89	16	11,34
Gammaridae	20	82,67	11,19
Capitellidae	48	0	8,4

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Discussion

The present *in situ* experiment showed a significant and diverse effect of the addiction of different types of detritus on infaunal and epifaunal assemblages.

The detritus loaded at the soft-bottom surface was rapidly degraded and assimilated into the benthic foodweb. This is not surprising, given that in sedimentary systems organic matter is rapidly decomposed (Vetter, 1996; Rossi, 2006).

This study provides evidences on the changes in the macrofauna assemblages associated to the deposition and degradation of detritus at small spatio-temporal scale. A recent review by Krumhansl & Scheibling (2012) described that the dynamics of detritivoures within mats of algae and sediments below it are largely dependent on the size and residence time of deposits, which in turn are determined by physical processes and the rate of consumption by herbivores. In cases of small-size mats and short residence times, diversity and abundance of macrofauna are locally enhanced relative to the surrounding sediments (Kelhaer & Levinton, 2003; Krumhansl & Scheibling, 2012 and references therein).

In fact, the detritus attracted a high number of amphipods in a very short term. However such attractive effect was confined to the upper part of the detritus and only few amphipods were collected in the sediment samples below each bag.

Lentdium mediterraneum was the dominant species in the soft-bottom assemblage. The abundance of *Lentidium mediterraneum* increased through times independently to the presence or types of detritus enrichment. The extreme abundance of this small bivalve, and his large spatial and temporal fluctuations, is typical of the benthic communities along the North-Western Adriatic shores (Ambrogi et al., 1995). However, as discussed by Bertasi et al. (2007) who worked previously in the same sedimentary environments studied here, the abundance of *L. mediterraneum* could not be consistently related to the direct or indirect effects of the defence structures.

The abundance and distribution of Capitellidae in the soft-bottom assemblage seemed to be affected by detritus 21 days after the start of the experimental detritus enrichment. The effect was detected only in the *Mytilus* treatment plots, but not in *Ulva* plots, possibly because of the reduced thickness of the *Ulva* mat at this sampling time. In fact, this opportunistic polychaete tolerates low oxygen and high sulphide conditions that can occur in high abundance beneath deposits (Levin & Smith, 1984; Thrush, 1986; Vetter, 1996; Okey, 2003; Bernardino et al., 2010).

In the detritus associated assemblage I found that, just after few days, *Mytilus* and *Ulva* attracted a huge quantity of amphipods, as well as their predators (i.e crabs belonging to Portunidae).

The abundance of amphipods attracted by both detritus types doubled at the second sampling date. Specifically, some of the major changes were related to few families of amphipods that showed the highest abundance in *Ulva* treatments after 7 days from the start of the experiment. This pattern seems reasonably related to habitat heterogeneity and food availability provided by detritus that could justify high occurrence of organisms from surrounding areas. Algal deposits attract assemblages of herbivores, which in some locations consume most of the detritus within days (Wernberg et al., 2006). Previous work has demonstrated that macroalgae in soft-sediment communities can serve as a refuge from predation for mobile epibenthic species (See Everett, 1994 and references therein). Furthermore, amphipods use mussel clumps as crevices, in order to found refuge from predators (Conlan, 1994).

After 21 days from the start of the experiment, the abundance of amphipods showed a dramatic decrease compared to the previous sampling date. This reduction was mostly related to a decrease in the wet weight of *Ulva*. However even after 21 days *Ulva* still maintain higher abundance of amphipods than *Mytilus* with the exception of the family Haustoridae that showed the highest abundance in *Mytilus*. Such family was not found during the previous sampling dates. Amphipods represent the diet of fish and crabs associated to algae beds (Dubiaski-Silva & Masunari, 2008). In particular few studies revealed that Dexaminidae were among the most abundant organisms found in the guts of both fish and crabs, which in turn may be attracted by high abundances of amphipods (Dubiaski-Silva & Masunari, 2008).

In summary, detritus originated from *Ulva* and *Mytilus* had significant and different effects on macrofauna. These effects included both changes in the composition and distribution of the macrofauna in the soft-bottom and effects related to the colonization of detritus by assemblages (particularly amphipods). These effects, however, were variable across the three sampling occasions. In the case of macrofauna associated to the soft-sediment, strong effects were evident 2 and 7 days after the beginning of the experiment, but weakened by 21 days. This suggests very rapid breakdown and assimilation of this material into the benthic foodweb. In the macrofauna directly associated to the detritus some of the major changes were related to few families of amphipods that showed the highest abundance in *Ulva* treatments after 7 days of the start of the experiment.

In conclusion the results of the present study suggests that mounds of detritus produced by organisms associated to artificial coastal defences can affect the structure of native soft-bottom assemblages. This phenomenon is particularly worrying if we consider that in the North Adriatic Sea, marked eutrophication processes, frequent storm events and high temperature could enhance detritus deposition around more than 190 km of coastline protected by artificial structures. This effect could become a serious environmental threat if scaled up to hundreds of km of structures as typical of many coastal areas in the world.

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CHAPTER 5

Effects of flooding in estuaries: an experimental comparison of detritus decomposition at different salinity regimes

Introduction

Coastal ecosystems, including estuaries and salt marshes, face threats from various environmental stressors potentially associated with global climate change (Thompson et al., 2002; Harley et al., 2006; Airoldi & Beck, 2007; Cardoso et al., 2008). Specifically, significant rise in sea level increase the likelihood of flooding events that endanger the coastal environments (Nicholls, 2004; Woth et al., 2006; De la Vega-Leinert & Nicholls, 2008). These impacts are particularly concerning given the predicted increase in the frequency and severity of extreme weather events (i.e. floods and storms) (IPCC, 2007). Given the growing risk and uncertainty generated by climate change, traditional coastal-defence structures can offer inadequate protection (Zanuttigh, 2011).

Over the past decades, European countries have invested a lot on maintaining and enhancing coastal-defence structures (Zanuttigh, 2011). Nevertheless, the high cost of maintenance, together with an increasingly holistic understanding of coastal processes (Nicholls, 2004), has led to the development of more sustainable strategies for coastal management (Evans et al., 2004). Integral to these new coastal management strategies is the appreciation of natural habitats as a key part of the coastal system (Living Shoreline Summit Steering Committee, 2006). Transitional habitats, such as salt marshes, are of immense environmental and economic importance, particularly for flood defence since their presence reduces the impact of waves at the shore line (Wolters et al., 2005). A salt-marsh restoration technique, called 'managed retreat' (or managed realignment), involves the tidal inundation of coastal land in front of re-aligned flood defences (Rupp-Armstrong & Nicholls, 2007).

Therefore, the development of alternative strategies of artificial coastal defences (i.e. no defence option) includes the understanding of alterations in ecosystem functioning in a flooding scenario. In estuaries, the flooding caused by seawater inundation can differently affect habitats along a gradient from marine to freshwater.

In rivers and estuaries the functioning of aquatic ecosystems is largely dependent leading by decomposition of allochthonous organic matter, enabling the recycling of nutrients and

chemical elements, sustaining important food chains and primary production (Goñi et al., 2003). Detritus dynamics in estuaries is likely to be complex, given that these systems receive inputs of organic detritus from multiple sources, including allochthonous terrigenous materials exported from land by rivers, allochthonous marine materials brought via tidal action from the open sea and autochthonous materials produced by estuarine macrophytes (Goñi et al., 2003). Breakdown rates on these different materials are also likely to vary along the length of the salinity gradient in estuaries (Lopes et al., 2011) and are likely to be dependent on the source of origin of the material in question.

Climate-driven alterations to flow regimes and sea levels are likely to alter the functioning of detrital pathways in estuaries. Firstly, the deposition patterns of organic material may change. Detritus from marine sources could be moved further inland and upstream through catchments, whilst estuarine and marine systems could receive increased quantities of terrestrial leaf litter. The consequence of such alterations to detritus distribution could be that detritus processing is altered because of a mismatch between the salinity regime and the detritus present, leading to direct effects of changes in salinity on breakdown rates, or indirect effects of changes in the associated detritivore assemblage or a combination of both.

In the present study I explored the breakdown rates of detritus from different sources (terrestrial vegetation, saltmarshes or macroalgae) across the freshwater-marine gradient in two estuaries located in South-West England. Specifically, I tested the hypothesis that each detritus type would decompose at the highest rate in the conditions of its native habitat, i.e. terrestrial detritus would decompose the fastest in freshwater conditions, saltmarsh detritus in brackish conditions and macroalgal detritus in seawater conditions. I also tested whether the macrofauna associated to each of the three detritus types would be affected by changes in habitat condition, from freshwater to brackish or seawater and viceversa.

Materials & Methods

Study sites

The experiment was carried out across the saline transition zone of the rivers Yealm (50°18.6'N, 04°4.2'W) and Erme (50°18.3'N, 03°57.0'W), two geomorphologically similar estuaries located in South Devon, UK (Fig. 5.1.). Both rivers rise on Dartmoor flowing for 16 and 20 km respectively before discharging into Wembury and Bigbury bays. Both estuaries are around 6 km long with a full salinity range from marine to freshwater. They

have similar catchment area (Yealm = 55 km^2, Erme = 43 km^2) and mean river flow discharge (Yealm = 1,7 m^3/s, Erme = 1,9 m^3/s) (Attrill et al., 2009; Sheehan et al., 2010). Because of the large tidal range (4,7 m) they can be classified as mesotidal (Davies, 1964). In both rivers, saltwater ingression into the freshwater zone is strongly limited by the presence of artificial weirs. Accepting that salinity is the main ecological factor defining estuarine gradients (Telesh & Khlebovich, 2010), I identified three salinity regimes (hereafter for simplicity habitats): freshwater (Fw), brackishwater (Br) and seawater (Sw). In both estuaries, the freshwater habitats were selected in the areas with predominant stream characteristics, located above the normal tidal limit (NTL) with salinity approximately zero throughout the tidal cycle. In the Yealm this habitat had a rocky to sandy substratum and a riparian community dominated by woodland (*Quercus*, *Alnus* etc.). The same habitat on the Erme (Fig. 5.2.) had a muddy-sand substratum, and the riparian community was dominated by grass pasture and scattered bankside trees (*Quercus*, *Alnus* etc.). The brackish habitats were located in areas equidistant between the NTL weirs and
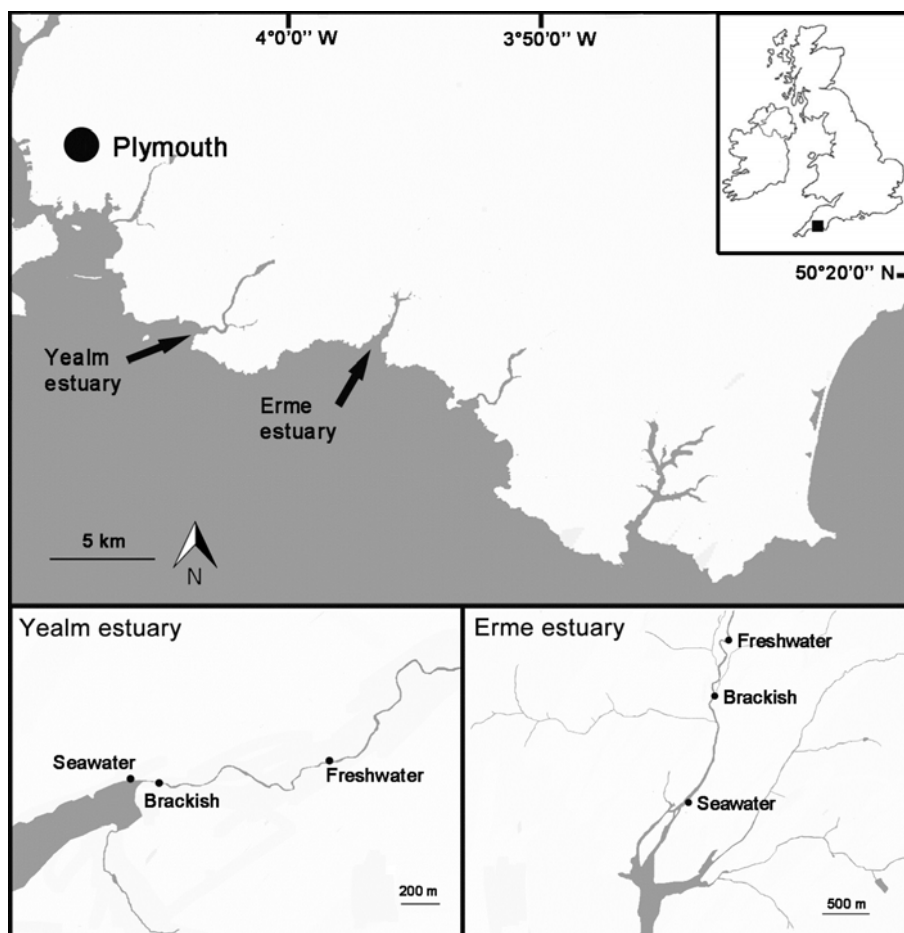
Fig. 5.1. Map of the study area, with indication of the three habitats where the experiments were carried out in each of the Yealm and Erme estuaries.

the open coast. In each of the two rivers, these areas had substratum characteristic similar to those found in the respective freshwater habitats. The vegetation in these habitats was characterised by the dominance of saltmarsh vegetation and ephemeral macroalgae (*Ulva* spp.). The seawater habitats were located at the mouth of the estuaries, in areas not impacted by boat traffic and breaking waves. In these areas, the dominant vegetation was fully marine, with the prevalence of complex macroalgae (e.g. *Fucus vesiculosus*). In the Erme the three habitats were located along 2 km stretch of estuary, whilst in the Yealm the passage from freshwater to seawater habitat occurred in less than 800 m (Rundle et al., 1998).

Experimental procedure

I studied the decomposition of two plants and one algal species characteristic of the three habitats: *Quercus robur*, Fagaceae leaves, typical of the freshwater habitats; *Spartina anglica*, Poaceae, typical of the brackish habitats, and *Fucus vesiculosus*, Fucaceae, typical of the seawater habitats. Leaf material was collected in May 2010 from adjacent woods (*Quercus*), salt marshes (*Spartina*), and the inter-tidal (*Fucus*) within the catchment of both rivers and oven-dried to constant weight (60°C for 72 hours).

Since detritus from the three sources has very different dry densities, I prepared litter bags (nylon cloth, 100 x 100 mm, 5 mm mesh size) with different weights but similar volumes in order to offer comparable surfaces for detritivore colonization. Air dried *Quercus* leaves were assembled in 5 g packs, *Spartina* leaves were cut into 8 cm long fragments (excluding the basal and apical parts) and assembled as 8 g packs, and *Fucus* detritus was assembled as 12 g bags.

Four replicate bags for each of the three litter species were deployed at each of the thre habitats (Fw, Br, Sw) at each of the two estuaries (Yealm, Erme). The bags were attached to ropes anchored to the river bed by bags of pebbles and steel pegs hammered into the sediment (Fig. 5.3.). The detritus was exposed in the field for 38 days based on degradation rates estimated from previous studies (Menéndez et al., 2001; Bärlocher, 2005; Sangiorgio et al., 2008; Sousa et al., 2008; Quintino et al., 2009). After this time the litter bags were retrieved and preserved in plastic bags containing 70% ethanol for subsequent analyses. Once in the laboratory, macro-invertebrates were extracted from the litter bags, identified at the lowest possible taxonomic level and counted. The detritus was washed to remove sediment, dried in an oven at 60°C for 72 hours and reweighed.

Fig. 5.2. Picture of the Erme estuary (UK).

Fig. 5.3. Litter bags submerged at the Erme estuary (UK).

Data analyses

Mass loss for each litter type was calculated as percentage according to the following equation: $\% L = (W_0-W_t)/W_0 \times 100$, where W_0 is the original dry weight of the litter and W_t was the dry weight remaining after 38 days. Differences in relative weight loss between litter species, habitats and estuaries were assessed via a three-way Analysis of Variance with four replicates for each factor combination. In the factorial design "Detritus" (De, three levels: *Quercus*, *Spartina* and *Fucus*) and "Habitat" (Ha, three levels: Fw - Freshwater, Br - Brackish, Sw - Seawater) were considered fixed factors, while Estuary (Es, two levels: Y – Yealm, E - Erme) was trated as random factor. ANOVA was carried out using SPSS v.18 package. Prior to ANOVA, the data were examined for normality and tested for homogeneity of variance using Levene's test and Arcsin(%) transformed where necessary. Tukey's HSD test was used to perform pairwise comparison for significant differences.

In order to provide comparable data comparable with other studies, the weight loss data were also modelled as decay exponential function $k = -(1/t) \times ln(W_t/W_0)$, (Olson, 1963). However, I did not test "k" by ANOVA as these data did not meet the assumption of homogeneity of variance.

The changes in the structure of macrofaunal, detritivore assemblages as a function of different detritus types, salinity and estuaries were assessed via a three-way Permutational Multivariate Analysis of Variance (PERMANOVA, Anderson et al., 2008) from the software PRIMER v6 (Clarke & Gorley, 2006), using the same logic as described previously.

Abundance data were log transformed to preserve information on relative covers of species, while reducing differences in scales among variables (Clarke & Warwick, 2001), and used to build a matrix of Bray-Curtis similarity coefficients. For the analysis, 9999 permutations of residuals under a reduced model were used to generate *P*-values (Anderson, 2001). For some terms in the analysis, there were not enough permutable units to get a reliable test by permutation, so a p-value was obtained using a Monte Carlo random sample from the asymptotic permutation distribution (Anderson & Robinson, 2003).

A non-metric multidimensional scaling (nMDS) ordination, calculated on the same Bray-Curtis similarity matrix, was used to visualize multivariate patterns of distribution of the macrofaunal assemblages in the experimental plots for each combinations of detritus type, salinity and estuary (Clarke & Warwick, 2001).

The similarity percentage routine (SIMPER) was used to highlight which taxa provided the largest contribution to dissimilarities between categories (Clarke & Warwick, 2001).

Results

Litter Breakdown

All litter bags were successfully recovered. Biomass loss through the 38 days of exposure varied considerably according to detritus type and position along the salinity gradient. Overall, the breakdown of *Quercus* litter was slowest, with weight loss never exceeding 42%, whereas *Fucus* litter decomposed fastest, up to 95% weight loss over the 38 days exposure (Fig. 5.4.). Each detritus type degraded at greatest rates in the habitat corresponding to their natural distribution (Fig. 5.4.). In particular, detritus from terrestrial vegetation and marine fucoid macroalgae had an opposite trend in breakdown rate along the freshwater-seawater gradient (Fig. 5.4. and Fig. 5.5.). Hence, biomass loss of *Quercus* litter significantly declined from freshwater (41,3 % ± 0,03) to brackish (28,4 % ± 0,04) and seawater (19,2 % ± 0,02) habitats (Table 5.1.). In contrast, the biomass loss of *Fucus* litter significantly increased from freshwater (80,5 % ± 0,02) to brackish (95,4 % ± 0,01) and seawater (95,2 % ± 0,01) habitats (Table 5.1.). Biomass loss for *Spartina* litter ranged from 61,8 % ± 0,03 in freshwater to 50,2 % ± 0,03 in seawater habitats, without consistent significant differences between habitats (Table 5.1.). There was also detected a significant

Table 5.1. ANOVA showing changes in dry weight loss (%) in relation to Detritus type (*Quercus, Spartina* and *Fucu*s, fixed factor), Habitat (Freshwater = Fr, Brackishwater = Br and Seawater = Sw, fixed factor), and Estuary (Yealm vs Erme, random factor). * = P < 0.05; ** = P < 0.01, ***=P<0.001, ns= not significant.

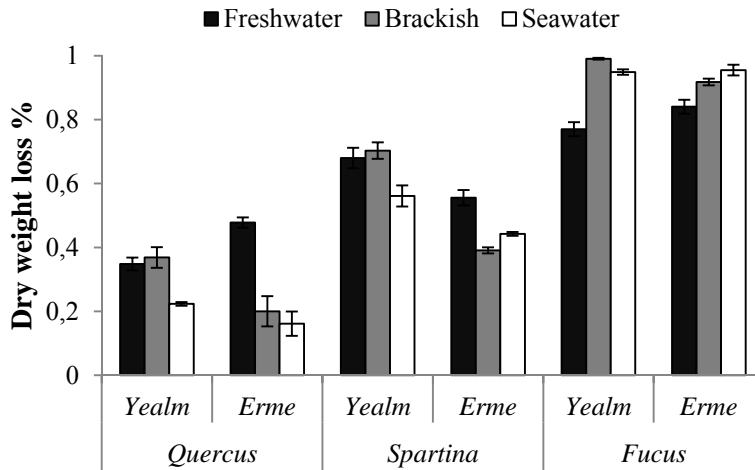| Source	df	MS	F	Pairwise comparisons	*Quercus*	*Spartina*	*Fucus*
De	2	4.717	63.721*	Fw *vs* Br	**	ns	***
Ha	2	0.026	0.164	Fw *vs* Sw	***	**	***
Es	1	0.174	0.789	Br *vs* Sw	*	ns	ns
De x Es	2	0.074	6.788				
Ha x Es	2	0.157	14.384*		Freshwater	Brackish	Seawater
De x Ha	4	0.226	20.738**	Erme *vs* Yealm	ns	***	ns
De x Ha x Es	4	0.011	2.094				
Residuals	54	0.005					

Fig. 5.4. Dry weight mass loss (% ± S.E., n=12, corresponding to 4 replicates for each detritus type) of overall detritus in the Yealm and Erme estuaries along the freshwater, brackish and seawater habitats.
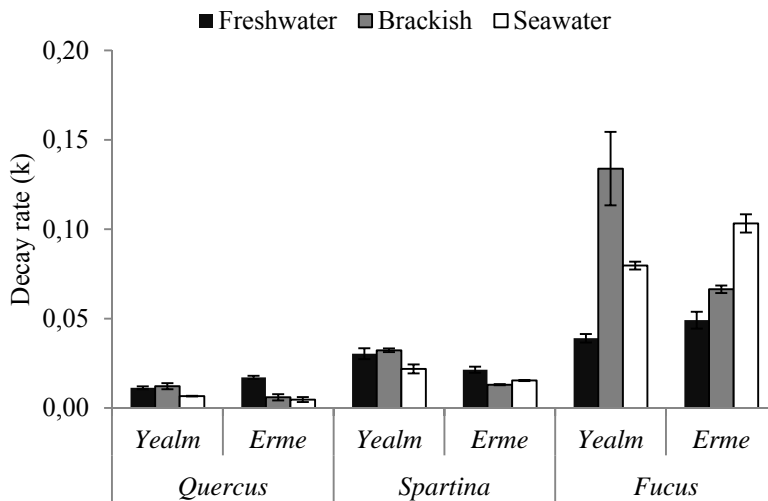

Fig. 5.5. Decay rate (k ± S.E., n=4 of *Quercus*, *Spartina* and *Fucus* in each of the freshwater, brackish and seawater habitats in each of the two study estuaries.
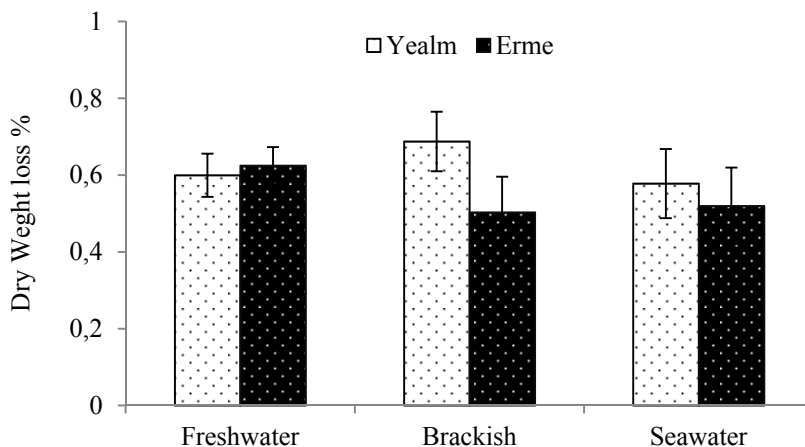

Fig. 5.6. Dry weight mass loss (% ± S.E., n=12, corresponding to 4 replicates for each detritus type) of overall detritus in the Yealm and Erme estuaries along the freshwater, brackish and seawater habitats.

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Habitat*Estuary interaction (Table 5.1.), indicating that while patterns of degradation in freshwater and seawater habitats were consistent between estuaries, degradation in brackish habitats was variable, with significantly greater degradation rates measured in the Yealm compared to the Erme (Table 5.2.; Fig. 5.6.).

Macro-faunal distribution

Thirty five species of macro-invertebrates were recorded. *Gammarus zaddachi* (Amphipoda) was the most abundant species (50,4 % of the total abundance) dominating all brackish and estuarine habitats. Chironomidae larvae (Diptera) (28, 6 %) were also numerous, particularly in freshwater sites. The third most abundant group were hydrobiid gastropods (7,3 %). Other common taxa included the juvenile crustaceans *Carcinus* (Decapoda) and *Jaera* (Isopoda) and the juvenile insects belonging to the families Leuctridae (Plecoptera), Ephemerellidae (Ephemeroptera) and Lepidosomatidae (Trichoptera).

Mean species richness of macrofauna associated to detritus decreased from freshwater (11,2 ± 1,4) to brackish (3,9 ± 1,4) and seawater (4,0 ± 0,9) habitats. This pattern was largely driven by the diversity of families of insects in the freshwater zone and the dominance of *Gammarus zaddachi* in the brackish and seawater habitats.

Table 5.2. PERMANOVA (35 variables, log-transformed data) showing changes in macrofaunal assemblages in relation to Detritus type (*Quercus, Spartina* and *Fucu*s, fixed factor), Habitat (Freshwater = Fr, Brackish = Br and Seawater = Sw, fixed factor), and Estuary (Yealm vs Erme, random factor). * = P < 0.05; ** = P < 0.01, ***=P<0.001, ns= not significant.

Source	df	MS	Pseudo-F	Pairwise comparisons	Freshwater	Brackish	Seawater
Es	1	17652	37,001***	*Quercus vs Spartina*	ns	ns	ns
De	2	3237,9	2,088	*Quercus vs Fucus*	ns	ns	ns
Ha	2	40553	3,8265	*Spartina vs Fucus*	ns	ns	ns
Es x De	2	1550,7	3,2505***				
Es x Ha	2	10598	22,215***		Erme	Yealm	
De x Ha	4	1573,1	2,8695*	Fw *vs* Sw	***	***	
Es x Ha x De	4	548,2	1,1491	Fw *vs* Br	***	***	
Resisuals	54	477,07		Br *vs* Sw	***	***	

There were also detected differences in numbers of individuals and in dominance patterns between the two estuaries. In the Yealm there were recorded 11,791 individuals, most of which were *Gammarus zaddachi* (63,1 %) and Chironomidae (21,2 %), whereas in the Erme there were collected only 4,358 individuals but with higher and lower representation of Chironomidae (48,8 %) and *Gammarus zaddachi* (16,1 %) respectively.

The multivariate analyses showed that the structure of macrofaunal assemblages differed both between habitats and estuaries (Fig. 5.7. and Table 5.2.).
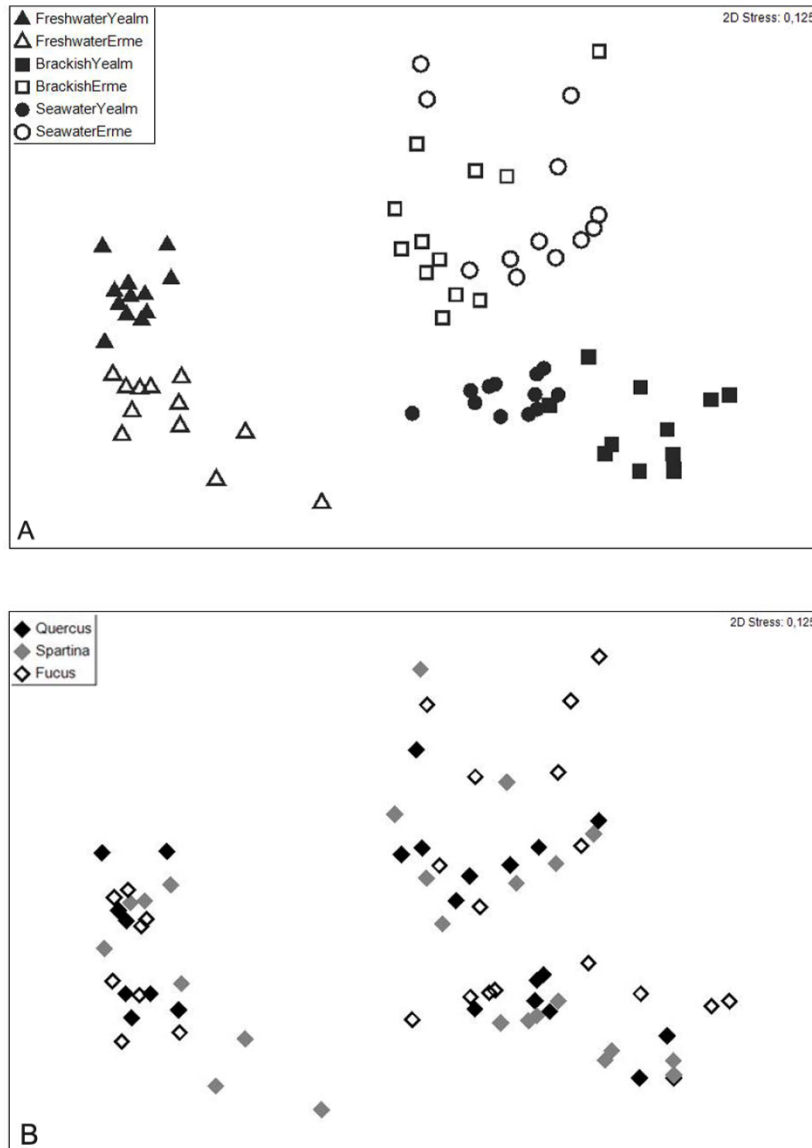
Fig. 5.7. nMDS ordinations for A) the combined factor Habitat-Estuary and B) for the factor Detritus.

Table 5.3. SIMPER showing the species most contributing to the Dissimilarities (Diss) detected in macrofaunal assemblages between different estuaries and habitats. J = Juvenile.

Species	Mean abundance		Diss/SD	% contribution to diss
	Erme	**Yealm**		
Average diss = 56,54				
Gammarus zaddachi	1,74	3,22	0,8	17,96
Hydrobiidae	1,56	1,25	1,25	13,1
Chironomidae (J)	3,14	2,99	1,1	11,97
Carcinus sp.	0,93	0	0,91	8,51
Jaera sp.	0,99	0,12	0,84	8,15
Carcinus sp. (J)	0,86	0	0,84	7,3
Oligochaeta	0,4	0,09	0,5	3,63
	Freshwater	**Brackish**		
Average diss = 85,85				
Gammarus zaddachi	0	3,91	2,04	14,2
Chironomidae (J)	4,9	1,57	1,86	12,57
Hydrobiidae	2,5	1,31	1,62	9,92
Leuctridae (J)	2,3	0	1,57	8,18
Ephemerellidae (J)	1,94	0	2,05	6,84
Lepidostomatidae (J)	1,96	0,07	2,02	6,84
Gammarus pulex	1,75	0	1,94	6,12
Elmidae (larvae)	1,61	0,03	1,84	5,7
	Freshwater	**Seawater**		
Average diss = 81,34				
Gammarus zaddachi	0	3,52	2,04	13
Hydrobiidae	2,5	0,4	1,43	8,75
Chironomidae (J)	4,9	2,74	1,73	8,49
Leuctridae (J)	2,3	0	1,62	8,31
Lepidostomatida (J)	1,96	0	2,15	7,34
Ephemerellidae (J)	1,94	0	2,11	7,05
Gammarus pulex	1,75	0	1,94	6,34
Elmidae (J)	1,61	0	1,93	5,91
Carcinus sp.	0	1,03	0,8	4,04
Asellus sp.	1,06	0	0,86	3,91
	Brackish	**Seawater**		
Average diss = 45,23				
Chironomidae (J)	1,57	2,74	1,22	27,77
Gammarus zaddachi	3,91	3,52	1,19	23,89
Hydrobiidae	1,31	0,4	0,81	9,79
Jaera sp.	1,03	0,63	0,88	8,85

The nMDS ordination in fig. 5a evidences how the assemblages are grouped on the base of the habitats and the estuaries. Furthermore invertebrate assemblage differed between habitats for each detritus type (De x Ha, p = 0,0456). The subsequent pair-wise tests revealed that in each habitat assemblages do not differ between detritus type. The very strong significant differences of the assemblages between detritus type among estuaries (De x Es, p = 0,0002) could be mainly attributable to differences between estuaries. Indeed "Detritus" as main factor resulted highly not significant (p(MC) = 0,1191). The nMDS in fig. 5.5.b shows that assemblages not clearly grouped on the base of different detritus type. The SIMPER analysis (Table 5.3.) showed that the freshwater habitats of the two estuaries were characterized by a greater presence of Chironomidae compared to both brackish and seawater habitats and by the almost exclusive occurrence of *Gammarus pulex* and of juveniles of taxa such as Leuctridae, Lepidostomatida, Ephemerellidae, Elmidae. Conversely, brackish and seawater habitats were dominated by *Gammarus zaddachi,* that was particularly abundant in the Yealm.

Discussion

The results of this study support the hypothesis that changes in detritus distribution and salinity regimes following flood events could alter normal processes of detritus decomposition in estuaries due to both direct effects of changes in salinity on breakdown rates, and indirect effects of changes in the associated detritivore assemblage. Indeed, I found that each detritus type decomposed at the highest rate in the conditions of salinity that were typical of its native habitat, and that the macrofauna associated to the detritus would be affected by changes in salinity, irrespective of the source of detritus.

Litter breakdown rates along the estuarine gradient varied according to their terrestrial or marine origin; oak litter decomposed faster in freshwater than seawater habitats whilst *Fucus* litter followed an opposite trend. Lopes et al. (2011) reported a similar pattern of decomposition for *F. vesiculosus* and the relatively rapid breakdown of *Fucus* litter in all environments is unsurprising given its low lignin and cellulose content and relatively high N-content compared to vascular plants (Tenore & Hanson, 1980).

The more fibrous leaves of oak have "slow" decomposition rates (Petersen & Cummins, 1974) and studies on *Quercus* litter breakdown in temperate freshwater ecosystems often report even slower decomposition rates than found here (Abelho, 2008; Castela et al., 2008; Lopez et al., 2001; Parkyn & Winterbourn, 1997; Molinero et al., 1996). Although

differences may be species-specific and linked to intrinsic factors such as nitrogen and polyphenolic content, environmental conditions may also play a part (Canhoto & Graça, 1996). However, there is a surprising paucity of literature detailing the breakdown of terrestrial detritus in brackish waters, although the present study with *Quercus* indicates that decomposition rates in brackish and seawater habitats are within the range of values observed in some freshwater studies.

Spartina litter is largely comprised of recalcitrant lignins (Lyons et al., 2010), but despite this *Spartina* spp. have a broad range of decomposition rates, influenced by position in the marsh and hydrological regime. In the present study the results in the Erme brackish site (k = 0,013) are similar to those reported by Sousa et al. (2008) for the Pancras saltmarsh in the Tagus estuary (Portugal) over a comparable time interval (k = 0,018; 31 days). Other studies have yielded faster decomposition rates for *Spartina alterniflora* in low marsh (Bouchard & Lefuevre, 2000 k = 0,028; Marinucci, 1982 k = 0,038), comparable to my results in the Yealm brackish habitat. I detected that differences in breakdown rates between brackish in the Erme and in the Yealm affected all three detritus types. However those differences are particular evident for *Spartina* which reached the highest value of weight loss in the brackish of the Yealm and lowest in the brackish of the Erme. In fact, like the others detritus type, *Spartina* degradead faster in the habitat of his natural distribution (i.e. the brackish-water), but this was true with regards of the Yealm estuary. This divergence might be related to different environmental conditions, specifically the lower oxygen concentration in the finer sediments of the Erme and the consequent limitation of degradation activity. Medeiros et al. (2009) reported in a laboratory study that hypoxic conditions slow down decomposition by aquatic hyphomycetes and it is also known that the hydrological regime plays a key role in the decomposition of plant detritus. In fact, water motion can influence decomposition process by its effects on mechanical breakdown, microbial colonization and oxygen concentration (Menéndez et al., 2001). Changes in decomposition process linked to shift in environmental conditions are better visualized if we include the analysis of the associated detritivore community. Although several studies have found that different litter types support different invertebrate assemblages (Wallace et al., 1982; Cummins et al., 1989; Graça, 2001; Leroy & Marks, 2006) I failed to detect any similar pattern here, instead finding strong differences in the structure and composition of macroinvertebrates assemblages between the freshwater and the brackish and seawater habitats. Nevertheless, the high rates of decomposition reported in this study for every detritus type, and the high abundance of certain detritivores, i.e.

chironomids in freshwater and gammarids in brackish and seawater regimes, may have neutralized major food preferences.

Community diversity, abundance, and dominance of certain taxa can all influence the processing rates of organic matter (Dangles & Malmqvist 2004; Cardinale et al. 2006; Abelho, 2008). Differences in macroinvertebrate composition, especially the leaf-shredding invertebrates, appeared thus to play a major role in breakdown rates among leaf species. The amphipod *Gammarus* is a highly opportunistic feeder (considered a facultative shredder by Cummins & Klug, 1979), but given the choice between different food items exhibits a certain degree of food selectivity (Friberg & Jacobsen, 1994). In the case of the present study, in the brackish and seawater habitats *Fucus* litter is colonized earlier and *Gammaruszaddachi* shift his colonization towards the harder leaves later on time.

The leathery leaves of *Quercus robur* were degraded faster in the freshwater because the ability of the shredders beloning to the families of Leuctridae, Leptoceridae, Limnephilidae and Sericostomatidae to successfully hydrolyzed and assimilate the refractory molecules of lignin, cellulose and hemicellulose.

In freshwater juveniles of Chironomidae often are the first invertebrate colonists on experimental leaf bags and reach densities far in excess of other taxa (Tank et al., 2010). There is a lack of knowledge on how non shredding taxa affect processing rates, although some evidence indicates that scrapers and gatherers facilitate the breakdown of organic matter when they are abundant. Canhoto and Graça (1999) showed that chironomids are able to mine the tougher leaves, attacking them from the inside and thus contributing to mass loss.

This study provides evidence that seawater inundations could lead to changes on detrital pathways in estuarine ecosystems. Specifically in brackish and marine waters decomposition of *Quercus* could be delayed because these habitats lack of a functionally diverse assemblage that includes shredders, scrapers, collectors and herbivores. Additionally freshwater habitats could receive an unusually high amount of marine derived detritus that will accumulate. Furthermore an increase of seawater intrusions could lead to a shift in detritivore community composition across salinity regimes, which may have consequences for organic material cycling in estuaries.

The flow of materials and organisms among habitats is often a key feature of population dynamics, energetics, and the structure of food webs and communities (Polis & Hurd, 1996). The predicted increase of seawater inundation into estuarine environments

(Nicholls, 2004) could have profound effects upon the natural distribution patterns of organic material along the marine - freshwater transition zone, with likely consequences for litter breakdown across environments, and the many decomposers that provide this essential ecosystem service (Polis et al., 1997; Costantini et al., 2009). The extent and nature of this impact depends both on the amount and origin of allochthonous material entering the estuary (Polis & Hurd, 1996) and the feeding activity and preferences of the decomposer community (Leroux & Loreau, 2008).

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CHAPTER 6

Conclusion

Urbanization is one of the major threats to biodiversity, because it is a driver for biotic homogenization (McKinney, 2006). In coastal areas urbanization is the result of multiple factors such as the pressure of human population, and the use of coastal resource and habitats for commercial purpose (Rebele, 1994). Moreover, the necessity to protect the land and the commercial activities and properties that insist along the coast increase the demand for active protection of the coastline from erosion processes and inundations. For this reason, the coastline in many areas of the world are armoured by seawalls and breakwaters, adding novel artificial substrata to coastal waters and replacing the natural substrates. The predicted increase of frequency and severity of storm surge and flooding together with the rise of sea level increases the necessity to protect the coast. However the evidence of the negative effects on biological communities and habitats urges for ecological-based adaptive strategies. These include the options of restoring the natural habitats and remove coastal protections to achieve a longer-term, more sustainable coastal management.

This thesis has analysed some of the ecological implications for coastal habitat and associated biological communities related to the choice of different strategies for coastal protection. In order to design effective management options, it is crucial to identify the criticism in the ecosystem. This thesis addresses some overlooked indirect ecological consequences from the use of artificial structures for coastal protection. It also explored some of the potential impact that the inundation could have in an estuarine environment in the absence of active defences.

Despite the threats of urbanization for biodiversity are becoming evident at global scales, urban ecology has so far focused mainly on terrestrial ecosystems (Shochat et al., 2006), while the consequences of urbanization of marine environment are poorly explored (Bulleri, 2006). In this context, it was not surprisingly the lack of any comprehensive information of the status of urbanization along the coasts of Sicily. The coastline of this region is heavily populated and exploited. Located between the western and the eastern basin of the Mediterranean sea, Sicily suffers an increasing proliferation of constructions

that border the coast. However no scientific information was available regarding the ecological implications of artificial structures, especially those regarding the hard coastal-defence structures. In the Chapter 2 of this thesis, the urbanization of the Sicilian coastline was quantified and characterized. Sicilian coastline resulted heavy urbanized as consequence of many structures for coastal protection, including the proliferation of industries, commercial activities and residential and tourist infrastructures that cover about 30 % of the overall coastline. In a management perspective, mapping and characterizing the state of urbanization of a specific stretch of coast is essential to address future options to mitigate the ecological impacts in a context of marine spatial planning. Focusing on the hard coastal-defence structures, an effective design and management plan should include the knowledge of the environmental and social context in which they are built (Zanuttigh, 2011). A future evolution of this work should focus on the elaboration of a GIS tool for a decision support system in coastal management that integrate information on the environmental, social and economic conditions. More effort is also needed to identify and quantify the site-specific impacts of coastal defence structures (Airoldi et al., 2005). In the Chapter 3 of this thesis I showed how the effects of artificial structures on the distribution and composition of biological communities can be related to the environmental context, specifically to the prevalent substrata surrounding the structure. In Sicily, fish assemblages use artificial structures differently, depending if the structure is located in a prevalent sandy area or in front of a rocky coast. Indeed, assemblages that populate natural rock reef were similar to those inhabiting artificial structures located in rocky areas, whereas they differed from those associated to artificial structure in sandy areas. Those differences were mainly driven by species with a benthic habit, probably because they are characterized by lower mobility and are likely to be more isolated from the nearby natural reefs.

This would allow monitoring the changes in the structure and composition of fish assemblages associated with artificial defence structures over time. Also expanding the work at different latitudes in the Mediterranean, would allow to explore possible relationship between the proliferation of coastal defences and the ongoing diffusion of thermophile species, including both native species ("meridionalization") and non-indigenous species ("tropicalization") (see Andaloro & Rinaldi, 1998; Azzurro, 2008).

Another heavy urbanized region along the Mediterranean sea is the North Adriatic coastline. In particular the west side of the north Adriatic is one of the most impacted coastline all over Europe (Airoldi et al., 2005). In fact the high concentration of human activities and proliferation of hard defence and other artificial structures in the sensitive

region of the North Adriatic led to interactive effects of coastal defence structures with other local impacts, such as regional eutrophication, recreational harvesting, local depletion of living resources, high rates of introduction of exotic species (Cencini, 1998; Correggiari et al., 1992; Airoldi et al., 2005; Bulleri et al., 2006).

Coastal defence structures built along prevailingly sedimentary coastlines such as those of the North Adriatic sea can affect surrounding soft-bottom environments and associated biota in several ways; for example, by habitat disruption, modification of water flow, sediment characteristics and detritus pathways, and predation by fish or other mobile predators (Airoldi et al., 2005). In the Chapter 4 of this thesis I experimentally manipulated the deposition of detritus detaching from breakwaters on the surrounding soft-bottom, and described the potential consequent changes in the native sedimentary fauna. Such effects of the organic enrichment on the infauna and epifauna occurred in a short time interval and could be site specific. More studies carried out in different defence systems are needed to generalize the results of the specific case of Lido di Dante presented in this thesis. In fact the extent of the modifications to surrounding habitat and biological communities is greatly influenced by the design of the structure. Structures allowing greater water flow from the seaward to the landward side could mitigate deposition of finer and organically enriched sediments, and help maintain similar habitat conditions and sediment characteristics at both sides of the structure (Martin et al., 2005). Confinement produced by the use of lateral groynes should be avoided to mitigate persistent accumulations of organic materials especially in the sheltered side of the structure, which could lead to stagnant water and greater biological oxygen demand (Airoldi et al., 2005).

In addition, could be interesting to explore the role of the organic detritus that accumulate around the defence structures in providing nursery area for juvenile fishes. In fact, fish abundance has been positively correlated with the presence and volume of detached macrophytes (Lenanton et al., 1982; Robertson & Lenanton, 1984; Lenanton & Caputi, 1989). Wrack accumulation provides food source and protection also for fish fauna (Robertson & Lenanton, 1984). In particular studies conducted by Martin et al. (2009) found that hard coastal-defence structures seem to provide habitats that appear to be suitable for new settlers, juvenile fish and other mobile fauna especially in the presence of accumulations of drifting algae. Furthermore since wrack can consist of different types of macrophytes (e.g. seagrass and brown algae), which exhibit different plant structure, fish may show a preference for wrack with different habitat structure (Crawley et al., 2006). Hence a future direction for experimental ecology could include the effect of different type

of detritus accumulated around the defence structures on the composition of fish assemblages, particularly juveniles.

Although in the Mediterranean artificial structures are the dominant measure adopted for coastal protection, in other parts of Europe alternative options are increasingly taken in consideration. In particular, the United Kingdom is at the fore-front in advancing knowledge for mitigation approaches and a more sustainable use of the coastal resources (Airoldi & Beck, 2007). In this context, coastal floods and erosion are seen as natural events, and an acceptable level of risk has to be taken in account (Zanuttigh, 2011). It is, therefore, essential to advance the knowledge of the environment vulnerability of those habitats at risk for seawater inundation, in order to proceed with no-defend options. The Chapter 5 of this thesis explored some potential ecological consequences of flooding in an estuarine environment. In the experiment carried out in southwest England, I found that estuarine ecosystems could be affected by the changes in the decomposition processes of organic material. In fact, if flooding alter the distribution of leaf-litter along an estuarine gradient, from near-shore to freshwater habitat, it would lead to a greater amount of algal material in the up river system that will accumulate, and an accumulation of terrestrial detritus in the estuarine-seawater habitat. Hence, the choice of "no defence" options must consider that the massive intrusion of seawater will affect the detrital pathways and cause shifts in detritivore assemblages. This study offer information on changes in decomposition rate and associated macrofauna in a mid-term time interval, such as that occurring during an extensive flooding. However, I suggest developing further research, by using manipulative field experiment to simulate short-term salinity shock such as occurring in storm surge events.

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