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Impact of coastal defence structures (tetrapods) on a demersal hard-bottom fish community in the southern North Sea

Stephanie Wehkamp*, Philipp Fischer

Biologische Anstalt Helgoland, Center for Scientific Diving, Alfred Wegener Institute for Polar and Marine Research, Kurpromenade 207, D 27498 Helgoland, Germany

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ABSTRACT

In the coming decades, artificial defence structures will increase in importance worldwide for the protection of coasts against the impacts of global warming. However, the ecological effects of such structures on the natural surroundings remain unclear. We investigated the impact of experimentally introduced tetrapod fields on the demersal fish community in a hard-bottom area in the southern North Sea. The results indicated a significant decrease in fish abundance in the surrounding area caused by migration effects towards the artificial structures. Diversity (HB) and evenness (E) values exhibited greater variation after the introduction of the tetrapods. Additionally, a distinct increase in young-of-the-year (YOY) fish was observed near the structures within the second year after introduction. We suggest that the availability of adequate refuges in combination with additional food resources provided by the artificial structures has a highly species-specific attraction effect. However, these findings also demonstrate that our knowledge regarding the impact of artificial structures on temperate fish communities is still too limited to truly understand the ecological processes that are initiated by the introduction of artificial structures. Long-term investigations and additional experimental *in situ* work worldwide will be indispensable for a full understanding of the mechanisms by which coastal defence structures interact with the coastal environment.

1. Introduction

In recent decades, global warming has resulted in changes in wind speed, wind direction and the frequency and intensity of strong storm events (Beaugrand, 2004; Wiltshire et al., 2010). Additionally, glacier melting is predicted to cause a rise in sea level of approximately 70 cm by 2100 (Hawkins et al., 2009; IPCC, 2007a, b). The consequences of these changes may include the destruction of coastal areas, including existing natural barriers, and the erosion of existing coastal protection measures through flooding and increasing high wave action (Airoldi et al., 2005a; Bacchiocchi and Airoldi, 2003; Moschella et al., 2005). Therefore, the establishment of additional coastal protection measures will become necessary in the coming decades to adapt to or minimise these effects (Bulleri and Chapman, 2010; Chapman and Underwood, 2011).

Until recently, most hard defence structures, such as concrete breakwaters and seawalls, have been placed directly in the intertidal area in front of the exposed coastline. However, such structures are also often introduced in the subtidal area at a distance from the coastline to absorb and dissipate wave energy (Charlier et al., 2005). This development in coastal protection strategies indicates that increasing subtidal coastal habitats will be altered through the establishment of artificial material.

As most artificial subtidal protection measures are permanent, future research should aim to quantify the ecological impacts of such anthropogenic structures whilst ensuring their engineering requirements with as little perturbation to the natural habitat as possible.

This approach is defined as ecological engineering (Browne and Chapman, 2011; Chapman and Underwood, 2011), and it addresses the commitments of the international community with respect to resilient management of coastal habitats as defined, e.g., by the guidelines of the Oslo and Paris Conventions (OSPAR, 1999).

In terrestrial environments, habitat loss caused by anthropogenic alterations is one of the most important factors in species decline worldwide (Sih et al., 2000). However, in aquatic environments, limited research has addressed the effects of habitat alteration on shorelines (Chapman and Underwood, 2011).

Often, it is unclear whether the impact of artificially introduced structures is “positive” or “negative” (Airoldi et al., 2005a; Moschella et al., 2005). The effects of artificially introduced structures are often

* Corresponding author. Tel.: +49 4725 8193357; fax: +49 4725 8193369.
E-mail addresses: stephanie.wehkamp@awi.de, stephanie.wehkamp@hotmail.de (S. Wehkamp).

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highly site specific and can vary over different spatial scales (Burt et al., 2009; Chapman, 2003; Clynick et al., 2008; Martin et al., 2005). The introduction of artificial hard substrata results in a change in the original habitat complexity (especially in soft-bottom areas) and consequently disturbs or changes the regional species community (for reviews, see Bulleri and Chapman, 2010; Chapman and Underwood, 2011). Artificial substrata can also serve as "stepping stones" for invasive species (Bulleri and Airoldi, 2005; Feary et al., 2011; Vaselli et al., 2008). One positive effect derived from artificial substrata could be the value enhancement of fish nursery grounds and fisheries, particularly in overfished areas (Cenci et al., 2011; Martin et al., 2005; Sayer et al., 2005).

In recent decades, studies have addressed the differences between natural substrata and coastal defence structures with respect to infaunal, epibiotic and invertebrate assemblages (Airoldi et al., 2005b; Ambrose and Anderson, 1990; Bacchiocchi and Airoldi, 2003; Browne and Chapman, 2011; Chapman, 2003; Martin et al., 2005; Wilding, 2006) as well as fish communities (Cenci et al., 2011; Clynick et al., 2008; Guidetti, 2004; Martin et al., 2005; Pizzolon et al., 2008; Wen et al., 2010).

Many studies have revealed significant differences between natural and artificial substrata with respect to fish abundance and richness (Arena et al., 2007; Clynick, 2006; Duffy-Anderson et al., 2003; Guidetti, 2004; Pizzolon et al., 2008) as well as species composition (Wen et al., 2010). Studies suggest that artificial substrata can act as a fish attractor and/or fish producer (Arena et al., 2007; Cenci et al., 2011; Guidetti, 2004; Simon et al., 2011). While many highly valuable studies have examined the effects of specifically designed artificial reefs on the enhancement of local fish communities (Bohnsack and Sutherland, 1985; Jensen, 2002; Seaman, 2007; Wilding and Sayer, 2002), only a few studies have examined the effects of coastal defence structures (Cenci et al., 2011; Clynick et al., 2007; Guidetti, 2004; Martin et al., 2005; Pizzolon et al., 2008; Wen et al., 2010). Most of the studied artificial reefs, however, were constructed completely or partially within coastal defence structures such as breakwaters (Bohnsack, 1989; Bohnsack and Sutherland, 1985; Feary et al., 2011; Froeschke et al., 2005; Pondella and Stephens, 1994).

Ecological studies on coastal defence structures are rare, particularly in northern Europe (Martin et al., 2005; Moschella et al., 2005), and there are almost no studies available that address their impact on fish (however, see Martin et al., 2005). Although 85% of the 1155-km North Sea coast of Germany is artificially protected (Rupp-Armstrong and Nicholls, 2007), to our knowledge, there are no published data available regarding the impact of coastal defence structures on the local fish community.

Previous studies have examined the impact of offshore wind turbines in the Baltic Sea (Wilhelmsson et al., 2006) and the North Sea (Reubens et al., 2011), wave power generators in Sweden (North Sea; Langhamer and Wilhelmsson, 2009) and oil rigs in Norway (North Sea; Jorgensen et al., 2002; Soldal et al., 2002) on fish populations. Furthermore, a few studies have examined artificial reefs in northern Europe (Jensen et al., 2000; Leewis and Halli, 2000; Sayer et al., 2005). All of these studies have found high abundances of fish either directly on or near the artificial substrata, and several studies have revealed differences among fish assemblages with respect to the natural surroundings.

This study aimed to reveal possible changes in diversity and abundance of the demersal fish community caused by the introduction of so-called "tetrapods" into a northern subtidal habitat. Tetrapods are artificial concrete structures that are used worldwide as breakwaters to absorb and dissipate wave energy either in the tidal zone or in the sublittoral zone in front of exposed hard-bottom coastlines (Gürer et al., 2005).

We established experimental tetrapod fields in a boreal hard-bottom area in the southern North Sea off Helgoland to evaluate the impact of coastal defence structures on the demersal fish community. Using fixed counting stations along line transects, we specifically analysed (1) differences in total and species-specific fish abundances before and after the introduction of the tetrapods, (2) spatial patterns in the demersal fish community with respect to the distance to the tetrapods and (3) the impact of the artificial structures on the main fish species in detail, including young-of-the-year (YOY) ages.

2. Materials and methods

2.1. Study site

Helgoland is located in the southern North Sea (German Bight) at 54°11′ N and 7°55′ E (approximately 50 km off the German coastline). The island is the tip of a 35-km^2 subtidal rock formation surrounded by the soft-bottom-dominated North Sea. This particularity led to geographic and ecological isolation from similar hard-bottom areas, the closest of which are located in Norway and Britain (Franke and Gutow, 2004).

The coastline of Helgoland is regularly impacted by heavy swells generated by storms, especially between September and March. Therefore, nearly the entire coastline is protected by artificial structures. Approximately 10,000 of these structures are tetrapods (four-footed concrete breakwaters each weighing 6 tonnes) (Fig. 1).

In February 2010, 3 experimental tetrapod fields were established north of Helgoland parallel to the shoreline and approximately 400 m from the coastline. The fields were introduced at 5- to 7-m depths based on the mean low-water spring (MLWS) according to hydrographic charts (Fig. 1). The 3 fields were placed 80 m from each other to reduce reciprocal impacts while ensuring similar environmental conditions. One field consisted of 6 tetrapods arranged in 2 rows of 3. The size of one experimental field was approximately 7 × 4.5 m, and the fields were approximately 2 m in height (Fig. 1). The area surrounding the tetrapod fields was dominated by rocky to pebbly substrate. A detailed study of the substratum typology of this area is provided in Wehkamp and Fischer (2012).

2.2. Survey methods

The assessment methodology was identical in 2009 (without tetrapods), 2010 and 2011 (with tetrapods). Visual census was conducted on SCUBA. Transect lines were laid out in a cross-like pattern (Fig. 1) at the beginning of the tetrapod fields (in 2010 and 2011) or at the proposed position of the tetrapod fields (in 2009), which was subsequently termed the 0-m sampling station. Sampling was performed every 5 m (0, 5, 10, 15 and 20 m) along each transect by counting and identifying all fish in a 1 × 1 m square to the left of the transect line and a 1 × 1 m square to the right of the transect line. The diver used a 1-m PVC rod to determine the size of the 2 × 1 m rectangle. At the 0-m sampling station, we counted the fish in a 1 × 1 m square to the immediate left and in a 1 × 1 m square to the immediate right of the tetrapod foot were the transect line was fixed.

Most of the target fish in the study area were relatively small (approximately 5–15 cm), highly camouflaged and remained mostly motionless on the ground or under and between stones. Therefore, we chose to use the two 1 × 1 m squares for each counting station. The advantage of this design was that fish could be detected successfully within these counting stations even under low visibility conditions, which were frequent. Furthermore, the diver had enough time to map one tetrapod field and the surrounding area without the risk of changing conditions.