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Assessing environmental effects of the bay scallop *Argopecten irradians* culture in China: Using abiotic and biotic indicators

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ABSTRACT

This study aims to identify the effect of biodeposition from a non-indigenous scallop (bay scallop) farm on sediment organisms in a semi-closed bay in Bohai Sea, China. Sediment characteristics, the meifauna community, and harpacticoid copepod assemblages were investigated before aquaculture and during the high biodeposition period at three scallop farms: Sandy-Shallow (SS), Muddy Shallow (MS), and Muddy Deep (MD), and their correspondent control sites. Total organic carbon and total nitrogen did not accumulate under the scallop farms. Although sedimentary bioavailable organic matter (biopolycarbon and Chlorophyll *a*) can accumulate under the scallop farms, all sediment characteristics were within the range of a meso-oligotrophic state. Bioindicators showed a location-specific response. Only in Farm SS the meiofauna community differed from the control site as there was a larger proportion of nematodes. The density and diversity of harpacticoid copepods were enhanced by scallop farms in MD, while a slight reduction of copepod density was found at SS due to the scallop farming. The overall results suggested that this non-indigenous scallop farming had minimal detrimental effect on the benthic environment in Laizhou Bay. Also, the bioavailable organic matter, the entire meiofauna community and harpacticoid copepod assemblages were useful tools for assessing of the potential environmental impact of shellfish farming

1. Introduction

The rapid expansion of marine aquaculture diminishes the tension between the demand and supply of seafood but also generates many environmental issues (FAO, 2016; Naylor et al., 2000). One of the most serious problems is the bottom-up impact on the surrounding ecosystem due to the releasing of abundant particulate or soluble organic materials to the sediment below (the so-called benthic environment). This can lead to organic accumulation on the sea bottom (Hargrave et al., 1997; Pearson and Black, 2000), affecting the cycles of sediment biogeochemistry (Chamberlain et al., 2001; Holmer et al., 2005), altering the diversity and the benthic community structure (Mirto et al., 2000; Netto and Valgas, 2010), and eventually impacting the overall ecosystem functioning (Callier et al., 2013; Dubois et al., 2007). Although marine shellfish aquaculture is considered to cause less environmental damage due to lack of feed loading, the high density of shellfish in aquaculture farms can enhance the downward flow of organic matter (Crawford et al., 2003). Many studies reported negative impacts of shellfish farming on the benthic environment (Chamberlain et al., 2001; Mirto et al., 2000; Stenton-Dozey et al., 2001), while others found the effects were too small to detect (Danovaro et al., 2004; Fabi et al., 2009; Han et al., 2013). So far, no general conclusion can be made and a better understanding of the environmental effects of shellfish farming is needed for its sustainable management.

China is the largest shellfish producer (FAO, 2016). Compared with other shellfish aquaculture, longline-suspended culture of scallops is relatively new and rapidly growing (Guo and Luo, 2016). The bay scallop (*Argopecten irradians* Lamarck) is a non-indigenous species (NIS) that was introduced from North America in 1982. Since then, bay scallop has dominated the scallop aquaculture, with a production of 0.817 million tonnes in 2012, which is still continuously expanding (Guo and Luo, 2016; MAC, 2013). NIS are popular in aquaculture in order to produce higher economic values because of their fast growth rates, wide range of diets, large tolerance to the environment, etc.

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(Ruesink et al., 2005; Silva et al., 2009). For example, bay scallop has a fast growth rate and reaches market size already within a year, which is twice as fast as native scallop species (Chinese scallop *Chlamys farreri*) (Guo and Luo, 2016; Xiao et al., 2005). However, NIS aquaculture involves ecological risks as species growing outside their native ranges can potentially cause ecological and economic harm to the environment (Olenin et al., 2007). In particular, introduced ecosystem engineers like bivalves could have more dramatic impact because their filter feeding and biodeposition activity can largely affect ecosystem structure and functioning (e.g. the biochemical fluxes and related organism communities) (Sousa et al., 2009; Zaiko et al., 2009). Higher biodeposition rates of bay scallop were reported in comparison to the Chinese scallop (Li et al., 2009; Wang, 2015; Zhou et al., 2006), but little is known about its impacts on the native communities, particularly the ones occurring in the sediment (so-called benthos or benthic community).

In view of the contrast between the ecological benefit provided by shellfish aquaculture and its potential environmental impact, systematic monitoring of the receiving ecosystem is urgently needed (Fabi et al., 2009), especially for the cases of cultured non-indigenous shellfish. Accordingly, identifying appropriate indicators of potential impact is one of the major tasks to monitor the shellfish aquaculture (Cranford et al., 2006). Traditional parameters like sediment characteristics are sometimes not sensitive enough to identify the biodeposition effect (Callier et al., 2008). Meiofauna is proposed as an integrative tool to monitor the organic pollution and recognized as an informative bioindicator (Grego et al., 2009; Mazzola et al., 2000; Mirto et al., 2012), because they respond to environmental stressors rapidly and their community structure gives information that cannot be detected with macrofauna (Giere, 2009; Kennedy and Jacoby, 1999; Semprucci et al., 2016). However, within meiofauna, the sensitivities and tolerances are variable among taxa (Raffaelli and Mason, 1981; Warwick, 1988). Harpacticoid copepods (Crustacea, Copepoda), the second most abundant taxon of meiofauna, are sensitive under the changing environmental conditions (De Troch et al., 2013; Giere, 2009; Wetzel et al., 2001). Also, they have an important function in the energy transfer from primary producers to higher trophic levels (Cnudde et al., 2015; Hicks and Coull, 1983; Leduc et al., 2009). Consequently, a possible community alteration of harpacticoid copepod could have a cascading effect on the food web.

The aims of this study are: (1) to evaluate the possible aquaculture impact of a non-indigenous scallop (bay scallop) on the benthic environment; (2) to assess the sensitivity and efficiency of indicators responding to biodeposition in the receiving benthic environment. We hypothesize that: (1) the presence of the bay scallop farm generates a negative effect on the benthic environment by changing sediment characteristics and modifying the structures of the meiofauna community and copepod assemblages; (2) Sediment characteristics (abiotic indicator), meiofauna community (rapid bioindicator), and copepod assemblages (lower-taxon bioindicator) provide different levels of information on the effect of bay scallop farming.

2. Material and methods

2.1. Study area and field sampling design

The bay scallop farming area is located in Laizhou Bay (east of Bohai Sea, Shandong Province, China; Fig. 1) and is one of the largest bay scallop marine aquaculture areas in China since 1980s, with a total area of 500 ha and an annual production of 144,470 t in 2011 (Li, 2013). Bay scallops are typically cultivated with the suspended long-line method. About 300 to 400 individuals of scallops are placed on a lantern net divided into around 10 cells by round plastic disks. 70–80 lantern nets are hung vertically on a suspended horizontal long-line, 5 to 15 m above the bottom and about 2 m under the seawater surface, and each farm has 10–150 lines parallel to and about 3 m apart from each other (see Han et al., 2013 for more details). Scallop farming activities are conducted from May to November each year without any provision of additional feed. Scallops are harvested when they reach a commercial size of 6 cm.

Three scallop farms, Farm SS (Sandy Shallow), Farm MS (Muddy Shallow), and Farm MD (Muddy Deep) located about 5 km, 10 km, and 20 km offshore the Laizhou coast, were investigated in May (i.e. before the scallop net are placed in this area) and in October 2016 (i.e. when the scallops reached heavy biodeposition rate, Wang, 2015). Correspondingly, three control sites (Ctrl SS, Ctrl MS, and Ctrl MD) were selected about 1.5–2.0 km away from each scallop farm to exclude any direct effect of farming (Fig. 1). Farm SS and Ctrl SS were located at a depth of 5 m above a sandy bottom (2% clay, 24–28% silt, and 69–73% sand). Farm MS & Ctrl MS and Farm MD & Ctrl MD were both located above the muddy bottom (4–6% clay, 53–55% silt and 39–42% sand) at depths of 7 m and 12 m, respectively. A northeast oriented dominant current with a mean velocity of 20 cm s^{-1} runs perpendicularly to the shoreline. Salinity and temperature of the water ranged between 30.7 and 31.1 and 20.4–21 °C in May 2016, and 29.8–30.1 and 11.4–12.2 °C in October 2016.

The sediment samples were collected by SCUBA diving using Plexiglass cores (3.6 cm inner diameter, 10 cm^2 surface area). Three deployments in each site were randomly chosen, and in each deployment three independent cores were obtained for meiofauna analysis, chlorophyll *a* (Chl *a*) contents, and other environmental variables including total organic carbon (TOC), total nitrogen (TN), biochemical composition of organic matter (i.e. carbohydrate, protein and lipid). For Chl *a* and other environmental variables, the top layer (0–1 cm) of the cores was sliced, homogenized and stored at −20 °C. For the meiofauna samples, the cores were sliced into three depth layers: 0–1 cm, 1–5 cm, and 5–10 cm (as in Hulings and Gray, 1971; Giere, 2009), and preserved in a 4% formaldehyde - filtered seawater solution.

2.2. Environmental variables of sediment

Chl *a* contents were measured in 2.5 g of wet sediments by fluorescence method after extraction with 90% acetone (Yentsch and Menzel, 1963). For the levels of TOC and TN, sediments were oven dried (60 °C), grounded, and treated with 1 N HCl for 24 h to remove carbonates. TOC and TN were measured with an Element Analyser Flash 2000 (Thermo Fisher Scientific). The C/N ratio was calculated from the TOC and TN levels. Sediment granulometry was measured by laser diffraction with a Malvern Mastersize 2000 particle analyzer (Malvern Instruments, UK).

The biochemical composition of the organic matter in the sediments (carbohydrate, protein, and lipid contents) was measured photometrically following the procedure of Fabiano and Danovaro (1994). About 0.2–0.3 g of dried and grounded sediments were used as a replicate for each type of analysis, and pre-combusted sediments (450 °C, 4 h) were used as blanks. Carbohydrate, protein and lipid contents were converted to the carbon equivalents by multiplying with the conversion factors of 0.40, 0.49, and 0.75 mg C mg^{-1}, respectively (Fabiano et al., 1995). Their sum was reported as biopolymeric carbon (BPC).

2.3. Meiofauna community structure and copepod assemblages

Sediment samples preserved in 4% formaldehyde were rinsed with filtered freshwater through 500 μm and 38 μm sieves. The fractions retained on a 38 μm sieve were centrifuged three times with Ludox HS40 (density: 1.18 g cm^{-3}) and stained with 0.5 g L^{-1} of Bengal Rose. Meiofauna was sorted to the higher taxon level. The first 100 adult harpacticoid copepods in each sample were picked with a needle, preserved in 75% ethanol, and identified to species level. Since the purpose of this study is to estimate the community structure and diversity, morphospecies were used. Only the results from the first sediment layer (0–1 cm) are reported in this study because a preliminary analysis has shown that > 80% of the meiofauna community was found