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Effects of the thermal discharge from an offshore power plant on plankton and macrobenthic communities in subtropical China

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ABSTRACT

The ecological impact of thermal discharge has become an important issue in the field of marine and environmental protection. We focused on the effects of thermal discharge on seawater temperature and biological communities based on data from before (2006) and after (2013–2014) the construction of a power plant. The thermal discharge induced stratification, which resulted in changes in the vertical hydrodynamic conditions. Stratification combined with elevated temperatures significantly affected the phytoplankton abundance and community structure. Elevated seawater temperatures decreased the chlorophyll-*a* concentrations by 34% and 63%, at the surface and bottom, respectively. The elevated seawater temperature at the bottom might not be high enough to significantly affect the macrobenthos, but significantly affected the phytoplankton and zooplankton communities. Because these communities serve as food for the macrobenthic community, their changes resulted in growth of the macrobenthos. Furthermore, this effect induced macrobenthic community succession, resulting in decreased species diversity and increased dominance.

1. Introduction

China's power generation capacity is 15.3 billion kWh and includes thermal power and nuclear power, which account for 65% and 2%, respectively, of the total power produced in 2015. According to China's 13th five-year plan (2016–2020), the nation's thermal power generation capacity will increase by 1.57 billion kWh and nuclear power generation capacity will increase by 0.29 billion kWh during the period covered by the plan. In recent years, thermal and nuclear power plants have been built to support social development. To satisfy the demands for water intake and thermal discharge, an increasing number of these plants have been built in coastal areas (Winter and Conner, 1978), which has brought the associated ecological impacts to the forefront of the field of marine and environmental protection (Shiers and Marks, 1973).

Elevated water temperature and residual chlorine (Cl) are the major threats posed by thermal discharge from coastal power plants, and previous studies have focused on their impacts on phytoplankton (Langford, 1990; Krishnakumar et al., 1991). Poornima et al. (2005) and Chuang et al. (2009) found the Cl introduced through thermal discharge to be the main reason of decreased phytoplankton abundance and chlorophyll-*a* concentration, particularly when the Cl concentration exceeded $0.2\,mg\,l^{-1}$. However, some studies (Briand, 1975; Li et al., 2014) have found that the elevation in water temperature due to thermal discharge, rather than Cl, is responsible for this change. Most of the previous studies (Brook and Baker, 1972; Briand, 1975; Li et al., 2014; Jiang et al., 2013) have observed negative impacts of elevated temperature on phytoplankton, but a few (Lo et al., 2004) have shown the impacts to be positive; the impacts include changes in both abundance and community structure (Martínez-Arroyo et al., 2000; Pane et al., 2001; Hutchins et al., 2007). For example, Hutchins et al. (2007) found that warmer water tends to limit the growth of more nutritious and palatable large diatom species while favoring that of less palatable and smaller diatom species and cyanobacteria. Previous studies on the impacts of thermal discharge from coastal power plants in China have focused on subtropical, semi-closed bays (Zeng, 2008; Li et al., 2014; Jiang et al., 2013; Tang et al., 2013) and similarly found that such discharge has negatively impact phytoplankton.

The macrobenthic community is considered a good indicator of ecological health (Smith and Simpson, 1995; Jewett et al., 1999) because changes in macrobenthic community composition, abundance and biomass can reflect the response of an ecosystem to human

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interference (Guidetti et al., 2000). Previous studies on the impacts of thermal discharge on the macrobenthos have been performed in rivers (Hogg et al., 1996; Durance and Ormerod, 2007; Worthington et al., 2015; Hu and Yang, 2001). When the temperature in a river is elevated by 2 °C–3.5 °C (or even by 1 °C in small rivers), reductions in the biomass and density of macrobenthic taxa have been observed (Hogg et al., 1996; Durrett and Ormerod, 2007; Worthington et al., 2015). In contrast, other studies have indicated that if the background water temperature is below 26 °C, a slight increase in the temperature increases the macrobenthic abundance (Hu and Yang, 2001).

Because water temperature increases due to efflux from coastal power stations far exceed 1.5 °C, stratification can be induced (Han, 2012; Tang et al., 2013). Roenunich and MdGowan (1995) and Zeng (2008) found that seawater stratification weakens wind-driven upwelling, thereby negatively influencing vertical nutrient exchange as well as phytoplankton and zooplankton growth. Previous studies of the ecological impacts of thermal discharge have focused on phytoplankton abundance and chlorophyll-*a* concentration at the surface. However, in addition to being negatively affected by the increased water temperature, bottom-dwelling phytoplankton and macrobenthos might also suffer from stratification-induced reductions in available nutrients.

In this study, we aimed to identify the effects of thermal discharge on bottom-dwelling plankton and macrobenthos and to determine whether stratification as well as elevated temperature affects plankton at the bottom and whether thermal discharge affects the macrobenthos through the ecological chain.

We chose a study area near the Ningde Nuclear Power Plant because the plant is the dominant human activity in this area, whereas other human activities have remained unchanged for the past ten years. A concentration of 0.2 mg l^{-1} has been demonstrated to be the critical Cl level for significantly suppressing phytoplankton productivity (Chuang et al., 2009). Compared with the threshold of 0.2 mg l^{-1} (Chuang et al., 2009) the Cl levels in the region near the power plant have been relatively low since the construction of the power plan, ranging from 0.01 mg l^{-1} to 0.03 mg l^{-1}, levels that may not be high enough to significantly affect phytoplankton. For this reason, only the effects of the temperature elevation caused by thermal discharge were analyzed in this study. Our objectives were (1) to ascertain whether elevated temperature induces stratification and, if so, to identify the thermocline based on vertical changes in water temperature in the impacted zone; (2) to evaluate the impacts of thermal discharge on the chlorophyll-*a* concentrations at the surface and the bottom of both the impacted and control zones as well as the plankton community structure, both before and after the impacts caused by the power plant (IPP); and (3) to determine the impacts of thermal discharge on the macrobenthic abundance and community structure by investigating the macrobenthic taxa, density and biomass before and after the IPP.

2. Materials and methods

2.1. Study area and sites

The study area (Fig. 1), which is adjacent to the outlet of the Ningde Nuclear Power Plant (26°56′N–27°5′N, 120°16′E–120°25′E), is located on northwestern Yushan Island in Ningde, Fujian Province, China. This area has a subtropical oceanic climate, and the mean annual temperature is 18.7 °C. The duration of the semidiurnal tides in the area near the outfall is regular. The directions of the flood and ebb tides are west-southwest and east-northeast, respectively. The plant consists of two units with an installed capacity of 1000 MW each; one unit began commercial operation in April 2013, and the other unit began operation in April 2014. The plant uses seawater as a coolant at a designed flow rate of 115 $m^3 s^{-1}$, and the cooling water is discharged into the East China Sea. The temperature of the cooling water in the outlet is 8.35 °C higher than that of the water in the inlet.

Our sampling strategy follows.

1) Eight cruises were conducted, four before and four after the IPP, the water sampling dates are shown in Table 1. Ecological sampling occurred in autumn from 10/7–10/2006 and on 11/6/2013, in winter from 1/1–3/2006 and on 1/16/2014, in spring from 4/1–8/2006 and on 5/13/2014, and in summer from 7/1–4/2006 and on 6/27/2014. Macrobenthos samples were collected only in autumn and spring.
2) Thirteen sampling stations with maximum depths ranging from 3.0 m to 15.5 m were established within 12 km of the outlet to avoid the effects of impingement and entrainment and to assess the thermal effect across the full range of conditions in the affected region. The sampling stations were divided into two groups (three impacted stations and 10 control stations) according to their distance from the outlet and the tidal current direction. The impacted zone includes Stations 1–3, which were established within 3.0 km north of the outlet, and the control zone contains Station 9 and Station 11, which were established 3.5 km southeast of the outlet, and Stations 4–8, Station 10, and Stations 12–13, which were located 5.3 km–12.7 km around the outlet.
3) Samples of the surface (depth of 0.5 m) and bottom (2.0 m from the bottom) phytoplankton communities were collected at each station to assess vertical variation. When the water depth was 5.0 m or deeper, both surface and bottom water samples were collected; otherwise, only surface water was collected.
4) Nine stations (T1–T9) were established around the impacted zone, and one station (C1) was established in the control zone to assess the vertical variation in seawater temperature in the impacted zone. The seawater temperature was investigated at six depth layers, specifically at 0.5 m, 1/5, 2/5, 3/5 and 4/5 of the total depth and 0.5 m above the bottom, during flood slack, ebb slack, maximum flood and maximum ebb on 6/5/2016.

2.2. Sampling methods

The water temperature (Temp) and salinity were monitored in situ using a thermometer and a salinometer, respectively. The dissolved oxygen (DO) was determined through Winkler titrations; the chemical oxygen demand (COD) was measured using the basic potassium permanganate method, and the five-day biological oxygen demand (BOD) was assessed via culturing for five days. To detect dissolved silicate (DSi), dissolved inorganic nitrogen (nitrate, nitrite and ammonium), dissolved phosphorus (DIP), and suspended solids (SS), the water samples were stored in 5-l buckets in the dark at 0 °C prior to analysis following the methods described by Jiang et al. (2012). The analytical methods used for the determination of turbidity (turbidit), Cl and boron (B) consisted of spectrophotometry, spectrophotometry with N,N-diethyl-1,4-phenylenediamine and spectrophotometry with turmeric, respectively; the samples were analyzed using a 7230 ultraviolet spectrophotometer. Sulfate was analyzed using the ion spectrometry method. Copper (Cu) was analyzed via flameless atomic absorption with a ZEEnit 600 atomic absorption spectrophotometer, and zinc (Zn) was analyzed by flame atomic absorption using a ZEEnit AA700 atomic absorption spectrophotometer. Mercury (Hg) was analyzed via atomic fluorescence spectrometry on an AFS-930 atomic fluorescence spectrometer.

The chlorophyll-*a* concentration was estimated after filtering a 2.5-l water sample obtained from either the surface or the bottom through a polycarbonate membrane. Pigments were extracted in the dark using 90% acetone (Sterman, 1988). The surface and bottom water samples were collected in 0.5-l bottles at each station, and all samples were stored in Lugol's solution. The samples for phytoplankton analysis were allowed to settle for 24 h; after sedimentation, the phytoplankton taxa were identified and counted using an inverted microscope.

Zooplankton were sampled from the bottom to the surface using a conical plankton net that was 50 cm in diameter and 145 cm long with a 505-μm mesh. The collected samples were preserved in 5% neutral