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Temporal pattern in the bloom-forming macroalgae *Chaetomorpha linum* and *Ulva pertusa* in seagrass beds, Swan Lake lagoon, North China

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ABSTRACT

Seagrasses that are distributed over a large area of the Swan Lake, Weihai, China, support a productive ecosystem. In recent years, however, frequent macroalgal blooms have changed the ecosystem structure and threatened the seagrasses. To understand the bloom-forming macroalgae we conducted a yearly field survey of Swan Lake. Results indicated that the macroalgae *Chaetomorpha linum* and *Ulva pertusa* both exhibited a much higher productivity and attained a greater maximum biomass (of 1712 ± 780 g DW m^{-2} and 1511 ± 555 g DW m^{-2}, respectively) than was the case for the seagrasses. The mean annual atomic ratios of *C*/*N*, *C*/*P* and *N*/*P* in *C. linum* were 14.31 ± 4.45, 402.82 ± 130.25, and 28.12 ± 2.08, respectively. The $\delta^{15}N$ values (11.09 ± 0.91‰ for *C. linum*; 9.27 ± 2.83‰ for *U. pertusa*) indicated a land-based source of *N* enrichment to the macroalgal blooms. High concentrations of nitrogen and phosphorus in the lagoon, particularly near the river mouth, supported the blooms.

1. Introduction

Anthropogenic influences, such as mechanical damage (associated with dredging, fishing, and anchoring), coastal construction, eutrophication, and biological invasions (Short et al., 1995; Hemminga and Duarte, 2000; Duarte, 2002; Orth et al., 2006; Williams, 2007; Short et al., 2014) are considered to be the most important factors contributing to the worldwide loss of seagrasses (Short and Wyllie-Echeverria, 1996; Waycott et al., 2009; Short et al., 2011). These factors stimulate the growth of, and therefore competition from, macroalgae (seaweeds), which has a negative impact on seagrass habitats, thus accelerating the decline of seagrass populations (Valiela et al., 1997; McGlathery, 2001; Baden et al., 2003; Holmer and Nielsen, 2007; Sugimoto et al., 2007; Huntington and Boyer, 2008). For this reason, the attention of researchers has focused on the interaction between macroalgal blooms and seagrass populations (e.g. Holmer and Nielsen, 2007) or coral reefs as well as its harm to human economies based on tourism, aquaculture or fisheries (Smetacek and Zingone, 2013).

In comparison with seagrasses, macroalgae are able to attain higher productivity and biomass, due to their direct response to increased nutrient levels. These “nuisance” macroalgal blooms usually manifest as a small number of “green-tide” species – for example, *Ulva*, *Cladophora*, *Chaetomorpha* and *Gracilaria* (Valiela et al., 1997; Guidone and Thornber, 2013) – that can out-compete seagrass species, resulting in the development of extensive, thick, and often-monospecific, drifting mats that float above seagrasses, or sink to a depth just above the sediment surface (Sfriso et al., 1993; Valiela et al., 1997; Hauxwell et al., 2001; Boese and Robbins, 2008).

The accumulation of high-density drift algae affects rooted vegetation as well as its associated fauna (Morand and Merceron, 2005; Holmer and Nielsen, 2007; Wennhage and Pihl, 2007; Sugimoto et al., 2007; Thomsen et al., 2012). It has been proposed that eutrophic conditions, which result in the growth of macroalgal canopies, result in the shading and subsequent decline of seagrass populations (Burkholder et al., 2007). Algal blooms can shade seagrasses, hindering shoot growth and survival (Hauxwell et al., 2001; Havens et al., 2001; Brun et al., 2003; Huntington and Boyer, 2008). In addition, the combined effects of the anoxic and high-sulfide conditions in the rhizosphere also contribute to seagrass die-off events (Carlson et al., 1994; Greve et al., 2003). Dense algal canopies reduce dissolved oxygen during decomposition (Koch et al., 1990), leading to a decrease in redox potential and an increase in the concentrations of sediment sulfide and toxic ammonium

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(Van Katwijk et al., 1997; Terrados et al., 1999; Lamote and Dunton, 2006).

There is however some evidence to suggest that the effects of macroalgae on seagrasses cannot be understood in isolation. Macroalgae mats may exert a negative, neutral, or potentially-positive effect, depending on interaction dynamics through time and space (Armitage et al., 2005; Klemens Eriksson et al., 2007; Jorgensen et al., 2010). Certain factors associated with the unique characteristics of particular sites (Hessing-Lewis et al., 2011), including plant size and the taxonomy of different seagrass species (Irlandi et al., 2004; Thomsen et al., 2012), and the diameter, height and longevity of macroalgal mats (Holmquist, 1997; Hauxwell et al., 2001; Irlandi et al., 2004; Huntington and Boyer, 2008), all play a role in determining the level of impact of macroalgae on seagrasses. Thomsen et al. (2012) concluded that unattached seaweeds have a greater negative impact on seagrasses than "rooted" seaweeds, and small seagrass species are more susceptible to such seaweeds than larger species.

Swan Lake is a coastal lagoon in North China characterized by the presence of large populations of *Zostera marina* and *Z. japonica* (Zhou et al., 2014b). The lagoon provides suitable habitats for bivalve mollusks, sea cucumbers and fish, which are found in great abundance in the Lake (e.g. Liu et al., 2013). Bivalve mollusks, as suspension feeders, are considered to be important components in coastal ecosystems, including seagrass systems (Newell, 2004; Zhou et al., 2006a; Booth and Heck, 2009). High population densities of two bivalve species, the Manila clam *Ruditapes philippinarum* and the Pacific oyster *Crassostrea gigas* – have been observed in the seagrass beds of Swan Lake. These two clam species may play an important role in the functioning of the seagrass ecosystems (Zhou et al., unpublished materials). In recent years, thick *C. linum* mats have covered almost the whole northern part of the lagoon, displacing seagrasses, with a resultant die-off of a large number of benthic organisms in the lake. During the early 1980s, a dam was constructed at the lagoon inlet/outlet, to prevent the escape of cultured sea cucumber to the open sea. Unfortunately, however, the dam prevented water exchange between the lagoon and the open sea, which had a severe negative impact on the lagoon ecosystem. In an attempt to restore the ecosystem, the dam wall was destroyed (by explosion), which resulted in a recovery of the seagrass and the return of swans to the lagoon. In recent years, however, a considerable growth of *C. linum* mats has been noted, resulting in a situation where this species has now replaced seagrasses as the main primary producer in certain areas of the lagoon. This has had a serious negative impact on some economically-important species in the lagoon. Recent blooms of *Ulva pertusa* have also been observed in the lagoon.

The aim of this study was to understand the distribution, growth cycle, and biomass of *C. linum* and *U. pertusa*, and thus to explore the possible causes of macroalgal blooms. Since the macroalgal mats mainly occur in the northern part of the lagoon, at the inlet of a river that carries a considerable amount of wastewater discharge. We hypothesize that macroalgal blooms are a result of the inflow of nutrient-enriched river water. The purpose of this project is to provide a scientific foundation for controlling macroalgal blooms, in order to facilitate seagrass recovery in this lagoon.

2. Materials and methods

2.1. Study area

The marine lagoon of Swan Lake (37°21′N, 122°34′E; Fig. 1), also called Moon Lake, is situated in Weihai, North China. The lagoon is connected to the Rongcheng Bay of the Yellow Sea via one outlet. Chengshan River is the main influent river, flowing southwards into this lagoon. Swan Lake has an area of 4.8 km^2 and the average depth is <2 m.

2.2. Sampling method and data analysis

During the period from October, 2011 to October, 2012, we conducted field investigations and sample collections in Swan Lake during every spring tide period. Two survey sites were set up for macroalgae measurements: one (for collection of *C. linum*) in the northern section of the lagoon and the other (for collection of *U. pertusa*) in the mid-eastern section (Fig. 1) in waters where the average depths of water at low tide are 0.3 m and 0.1 m, respectively. During low tide, the intertidal beds could be accessed for a sufficiently long period to enable the collection of samples from the two survey sites. A 30 cm × 30 cm quadrat was used to measure the biomass of *C. linum* and *U. pertusa* within their distribution sites, based on five to six randomly-selected replicates (within ca a 100-m^2 area). The number and height of seagrasses in each quadrat, if found, were measured *in situ*. Macroalgae samples, collected from each quadrat, were individually placed in plastic bags and transported to the laboratory while being maintained in dark, low-temperature conditions. All samples were rinsed with fresh water, after which the wet weight and dry weight (dried at 80 °C to constant weight) were determined. To determine the levels of *C*, *N* and *P*, triplicate samples (ca. 200 g) of *C. linum* and *U. pertusa* were randomly collected during each survey. All samples were rinsed with fresh water and dried (at 60 °C), then ground to a fine powder using a pestle and mortar. *C* and *N* contents were determined using a VarioEL III CHONS analyser. A modification of the method developed by Solórzano and Sharp (1980) was used for the determination of particulate P and total P (Zhou et al., 2003). The distribution areas of *C. linum* and *U. pertusa* were determined by means of GPS readings taken from a boat traveling along the edge of the *C. linum* mats. Water temperature and salinity (YSI monitor) were measured at high tide in the center of Swan Lake (L7; Fig. 1).

Three replicate sediment samples, at a depth of 5 cm, were taken on a monthly basis, in the area where *C. linum* occurred. The sediment samples were dried at 60 °C and then crushed to a fine powder. Subsamples were treated with HCl vapor for 16 h, to remove carbonates prior to organic carbon (OC) and total nitrogen (TN) analysis, with a VARIO EL III elemental analyzer standardized with acetanilide. Subsamples were also analyzed for total phosphorus (TP) and organic phosphorus (OP), using a modification of the method developed by Solórzano and Sharp (1980) for the determination of particulate P and total P (Zhou et al., 2003). Triple-replicate samples of sediment (to a depth of 15 cm in sediment) were also collected on 18 October 2012, for grain size determination, based on laser diffraction analysis, sieve analysis, and comprehensive analysis.

To assess the impacts of river-water input, we monitored water temperature and salinity on 18–20 August 2012 along a distance gradient from the mouth of Chengshan River to the center of the lagoon (Fig. 1). Ten survey sites were set up, with three sites (R1–R3) located in the river and the other seven sites (from L1 to L7) located in the lagoon. Average depths of water in the lagoon sites L1–L7, compared to the lowest tidal level were 0.2, 0.3, 0.0, 0.2, 0.3, 0.5, and 0.8 m, respectively. Sites R1–R3 were all at a depth of 0.2 m. During the survey period, monitoring was conducted each day at low tide (between 18:00 and 20:00) and at the high tide (between 11:00 and 13:00). At each of the survey sites, 50 ml of replicate surface water samples were collected and cryopreserved as soon as possible. An automatic nutrient analyzer was used for the determination of NH_4–N, NO_3–N, NO_2–N, and PO_4–P.

To assess the impacts of macroalgal blooms on the economically important benthic mollusks, i.e. the Manila clam *R. philippinarum*