

Zooplankton community in Yangtze River Estuary and adjacent sea areas after the impoundment of the Three Gorges Reservoir

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Received 31 January 2016; Accepted 9 May 2016

Abstract – Zooplankton is an important component of aquatic ecosystems and community variation reflects environmental changes. The impoundment of the Three Gorges Reservoir (TGR) has dramatically altered hydro-environmental conditions from hydrological regimes to water quality, which may have a serious impact on the estuarine ecosystem. This study analyzed the characteristics of the zooplankton community as well as their relationships to hydro-environmental factors in the Yangtze River Estuary (YRE) and the adjacent sea areas using the data after TGR impoundment. The historical zooplankton data over the past 30 years were also collected from the literature to explore long-term changes of the zooplankton community by the Mann–Kendall test. The results show that the dominant species were *Noctiluca scintillans* and *Calanus sinicus* in spring, *Acartia pacifica* in summer, and *Paracalanus aculeatus* in autumn. The most important environmental factors affecting the zooplankton community for the whole year were salinity and temperature of the bottom layer, which were largely influenced by river discharge and thus by reservoir regulation. Combined with the long-term historical data, the biomass and population density of zooplankton showed an increasing trend, and the community structure of zooplankton also shifted over the past 30 years.

Key words: community structure / spatial-temporal variations / bottom layer salinity and temperature / changing trend

Introduction

Zooplankton play a pivotal role in marine food webs by transferring carbon fixed by phytoplankton to higher trophic levels, and are commonly used as a food proxy for fish larvae or larger invertebrates (Hughes, 2000). In addition, as most zooplankton species are not commercially exploited, their abundance variations are mainly related to hydro-environmental changes. As a result, they are often used as indicators of the ecosystem state (Hardman-Mountford, 2000; Hays *et al.*, 2005). Moreover, zooplankton responses can amplify subtle environmental variations that are difficult to detect by assessing single physical variables (Fernández-Urruzola *et al.*, 2014). A large number of studies have highlighted how plankton (particularly zooplankton) may be an important indicator of changes in marine ecosystems, because they are not commercially

exploited and are short-lived (Taylor *et al.*, 2002; David *et al.*, 2005; Primo *et al.*, 2009; Mialet *et al.*, 2011; Honggang *et al.*, 2012).

Situated at the interface between the continental and oceanic domains, estuaries are subjected to frequent and significant environmental changes. In particular, the chemical composition of estuarine waters is determined by highly variable fluvial flows (Lam-Hoai *et al.*, 2006). Over the last 50 years, fluvial flow patterns and sediment have been greatly modified by river dams (Dai and Liu, 2013; Dhivert *et al.*, 2015). It has been reported that more than 45 000 large dams have been constructed around the world, with China ranking first with 22 265 dams, accounting for 44.80% (Chen *et al.*, 2011). Consequently, around 70% of the world's rivers are intercepted by large reservoirs (Kummu and Varis, 2007). These changes in river flow regime can lead to alterations in the diversity of fauna and flora, and affect the ecosystem downstream (Li *et al.*, 2010; Chen *et al.*, 2011; Haghghi *et al.*, 2014).

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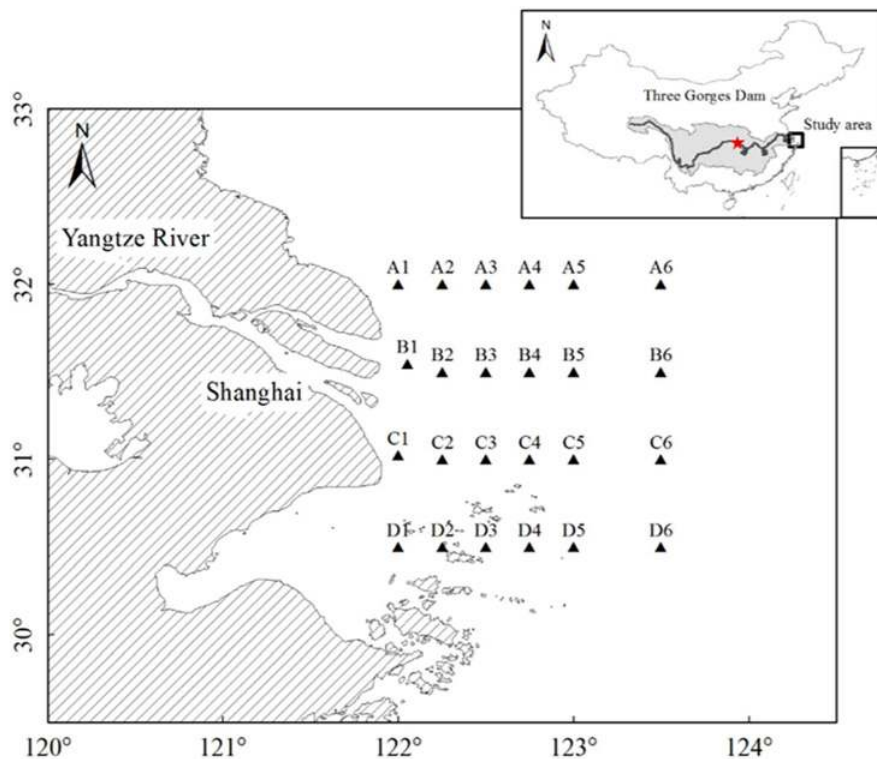


Fig. 1. The study area and sampling sites in the YRE and adjacent sea areas. Each dot represents a sampling site. The survey was conducted in August 2010, November 2010 and May 2011.

For instance, in the Egyptian coastal waters prior to the construction of the Aswan high dam, high nutrient concentrations in the Nile River produced a dramatic bloom of phytoplankton, which supported a productive fishery (El Bastawesy *et al.*, 2014). Due to the loss of over 90% of the freshwater flow, the phytoplankton bloom disappeared and the fisheries collapsed. Along the Mississippi River, numerous dams and extensive levee construction have resulted in a dramatic decrease in flooding and flux of sediment to low-lying regions (Milliman, 1997), leading to a rapid loss of coastal land by the expansion of coastal lagoons. In general, dam construction usually slows flow velocity, prolongs water residence time (Friedl and Wuest, 2002) and finally influences the structure of the aquatic ecosystem (Wu *et al.*, 2010).

The region of the Yangtze River Estuary (YRE) is an important industrial and economic center for China. In recent years, the hydrological, sediment and nutrient regime have been substantially modified due to massive construction of water works and agricultural and industrial developments in the upstream, causing severe eutrophication, frequent red tides and an expanding hypoxic zone (Chen *et al.*, 2012). The construction of the Three Gorges Reservoir (TGR) was intended to improve flood management and produce hydropower. It began to store water on June 1, 2003. However, the construction of the TGR has caused widespread scientific controversy and worldwide concerns because the project has been argued to lead to serious changes in hydro-environmental conditions, which may influence the ecosystem of the

downstream and the estuary. Since the 1950s, many investigations of zooplankton in the YRE have been conducted (Chen *et al.*, 1985, 1995; Xu *et al.*, 1995a, b, 1999, 2003; Guo *et al.*, 2003; Sun *et al.*, 2003). However, studies on zooplankton assemblage changes and their responses to hydro-environmental variations in the estuary before and after dam construction are scarce. We hypothesized that the changes of flow regime due to the TGR may alter the salinity and temperature conditions of the YRE, and thus affect the zooplankton community.

The objectives of this study were to (1) understand the composition and structure of zooplankton spatially after the TGR; (2) identify the most important hydro-environmental variables affecting the distribution of zooplankton species; and (3) analyze the annual variability of the zooplankton assemblage over the past 30 years and try to reveal the effects of the TGR on YRE zooplankton.

Methods

Study area

The YRE and adjacent area (Fig. 1, E122.0°–123.5°, N30.5°–32.0°) is located at the center of the eastern Chinese coast (Shi, 2004). The Yangtze River, which flows into the YRE, is the third longest (6300 km) and fifth largest river in terms of annual runoff ($900 \text{ km}^3 \cdot \text{y}^{-1}$) in the world, with a catchment area of about $1.8 \times 10^6 \text{ km}^2$ (Jiang *et al.*, 2014). The river catchment has the East

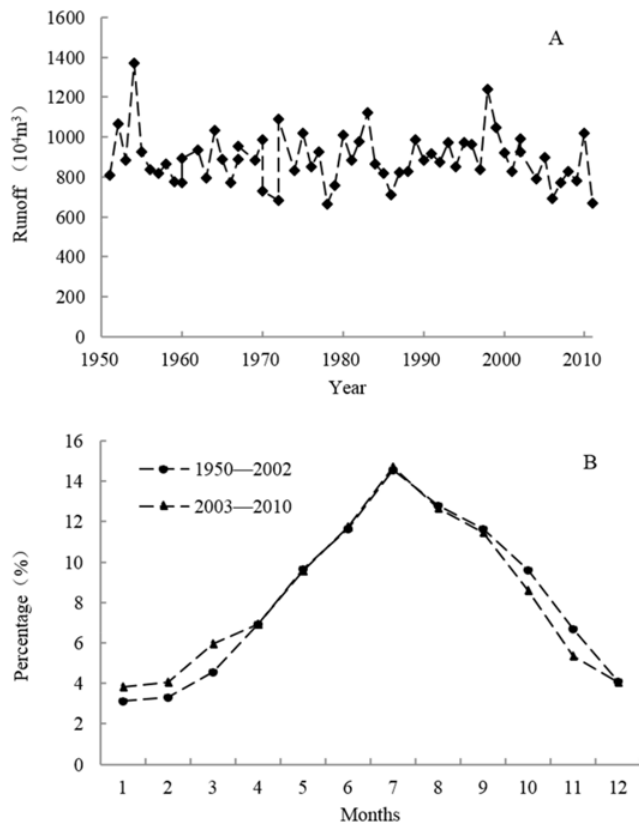


Fig. 2. The variation of annual and monthly discharge at the Datong station before and after the impoundment of the Three Gorges Reservoir. Percentages are the values that each month's discharge accounts for in the total discharge throughout the year (data from website: <http://www.cjw.gov.cn>).

Asian monsoonal climate. Water delivery accounts for 71.7% of the total annual discharge during wet seasons (May–October), and only 28.3% in dry seasons (November to the following April) (Xu *et al.*, 2009). Annual discharge has shown little variation since 1959 with an average annual runoff of $2.8 \times 10^4 \text{ m}^3 \cdot \text{s}^{-1}$ (Fig. 2). Sediment discharge and concentration has decreased significantly in the last decade (Dai *et al.*, 2011; Feng *et al.*, 2014). The Yangtze River basin, especially the lower reaches and the estuarine area, is characterized by high industrialization and urbanization. The nitrogen and phosphorus load in the YRE has increased approximately seven- to eight-fold since 1960 (Shen, 2001; Chen *et al.*, 2011); however, the dissolved Si concentration has decreased gradually since TGR impoundment (Chai *et al.*, 2009).

Sampling and laboratory analyses

Zooplankton and environmental data were collected from 24 stations in August 2010 (high-flow in summer), November 2010 (low-flow in autumn), and May 2011 (normal-flow in spring). The sampling sites are illustrated in Figure 1.

Table 1. List of historical data on zooplankton in the YRE from the literature.

Time	Study area	Source
1982	31–31°45'N, 120–122E	Zhu (1988)
1988–1989	30°50'–31°20'N, 122°20'–122°50'E	Chen (1995)
1999–2000	31°00'–31°32'N, 121°21'–122°30'E	Guo (2003)
2000–2003	30°–31°30'N, 121°30'–122°30'E	Xu (2005)
2002	30°40'–31°36'N, 121°23'–123°40'E	Ji (2006)
2003	30°40'–31°36'N, 121°23'–123°40'E	Liu (2007)
2004–2006	30°30'–32°N, 121°30'–123°E	Xu (2009)
2005	29°30'–32°N, 121°–123°E	Zhang (2009)
2009	29°54'–31°30'N, 121°–122°46'E	Zhu (2011)
2010–2011	30°30'–32°00'N, 122°00'–123°30'E	This study

At each site, three water samples were collected from the surface (0.2 m below water surface), middle and bottom (0.2 m above sea bed) of the water column. The maximum depth of sample sites was 60 m and its location was about 150 km off shore. Water samples were collected using 5.0 L Nisk water samplers, stored in a portable refrigerator at $< 4^\circ\text{C}$, and then analyzed for nutrients (silicate, total dissolved nitrogen, total dissolved phosphorus) and chlorophyll a (Chl-a) concentrations in the laboratory. Water samples for nutrient parameters and Chl-a were filtered through $0.45 \mu\text{m}$ cellulose acetate membranes and Whatman GF/F filters, respectively. The procedures of Grasshoff *et al.* (1983) and Zhang *et al.* (1997) were followed for nutrient concentration analysis, and the process of Lorenzen (1967) was followed for Chl-a analysis. To determine the total suspended solids (TSS), a known volume of well-mixed sample was filtered through a dried and weighed membrane filter with a pore size of $0.45 \mu\text{m}$. Water temperature, salinity and pH were measured with a CTD instrument (CTD 90 M, Germany) in the field.

Field sampling and laboratory analysis of zooplankton were conducted according to the Specifications for Oceanographic Survey (State Oceanic Administration, 1991). Zooplankton were collected with a shallow water type I plankton net (mouth size: 0.2 m^2 ; mesh size: $505 \mu\text{m}$) along the vertical direction from the seabed to the surface. The samples were preserved in 5% buffered formalin. At the laboratory, all zooplankton individuals were identified to the species level. Zooplankton individuals were counted to obtain the population density ($\text{ind} \cdot \text{m}^{-3}$) and wet weights were measured to obtain biomass ($\text{mg} \cdot \text{m}^{-3}$). The founded species were listed in Appendix.

Historical data collection

The historical data on zooplankton composition, population density and biomass over the past 30 years were collected from the published literature (Table 1) and were summarized. Since most research was performed in summer, only the data for the summer were used in this study, for consistency and comparable diversity, to detect the annual trends of zooplankton variation over the past 30 years through the Mann–Kendall (MK) test.

Considering the seasonal runoff characteristics of the Yangtze River, the period from June to August was considered as summer.

Data analysis method

Species dominance: Equation (1) was used to calculate the species dominance index:

$$Y = \frac{n_i}{N} f_i \tag{1}$$

where n_i is the number of individuals of species i ; f_i is the frequency of species i ; and N is the total number of zooplankton individuals. Species with a dominance index higher than 0.02 were taken as dominant species.

Relationships between zooplankton and hydro-environmental factors: Biota-Environment (BIOENV, Primer 5) was used to investigate the relationship between the zooplankton assemblage and hydro-environmental

factors. The Spearman correlation coefficients between the zooplankton assemblage and hydro-environmental factors were calculated. The population density data were transformed through $\log(x + 1)$ to balance the contributions from the few very abundant species with the many rare species (Clarke and Warwick, 2001).

Zooplankton annual variability: the MK test, a non-parametric method for detecting trends in time series, was used to examine annual zooplankton trends for the past 30 years. The test statistic Z indicates the presence of a trend, where a negative (positive) value of Z indicates a downward (upward) trend. In this study, the significance level α was set to 0.05 and 0.01, and the corresponding $Z_{1-\alpha/2}$ was 1.96 and 1.65, respectively. The MK test was conducted in R using the Kendall package (version 2.15.3; R Development Core Team, <http://www.r-project.org/>).

Results

Zooplankton community features

A total of 178 species were collected in all three seasons. Copepods (86 species) accounted for 48.31% of zooplankton assemblage richness, Medusa (17 species) accounted for 9.55%, Planktonic larvae (19 species) accounted for 10.67%, and the remaining groups accounted for less than 5%. In spring, 97 species were identified, including 33 Copepods, 13 Medusa and 46 other taxonomic species. In summer, there were 73 species of Copepods, 10 species of Medusa and 69 other taxonomic species. In autumn, 131 species were identified, including 65 species of Copepods, 9 species of Medusa and 57 other taxonomic species (Fig. 3).

Three dominant species in spring and five dominant species in both summer and autumn were identified (Table 2). *Macrura* larva was the most common dominant species in all seasons. Beside *Macrura* larva, *Acartia pacifica* and *Paracalanus aculeatus* were the common

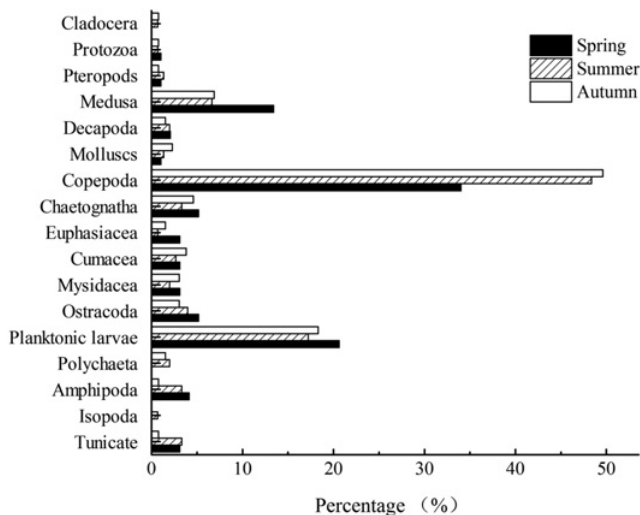


Fig. 3. Percentages of some identified zooplankton groups in the Yangtze River Estuary and adjacent sea areas in 2010–2011.

Table 2. Dominant species and dominance index of zooplankton in the Yangtze estuary.

Season	Number	Dominant species	Groups	Population density*	Dominance
Spring	1	<i>Noctiluca scintillans</i>	Protozoa	275.71	0.3822
	2	<i>Calanus sinicus</i>	Copepoda	233.69	0.3240
	3	<i>Macrura</i> larva	Planktonic larvae	89.62	0.0207
Summer	1	<i>Acartia pacifica</i>	Copepoda	147.22	0.2255
	2	<i>Macrura</i> larva	Planktonic larvae	62.78	0.0913
	3	<i>Evadne tergestina</i>	Cladocera	93.99	0.0864
	4	<i>Noctiluca scintillans</i>	Protozoa	75.19	0.0864
	5	<i>Paracalanus aculeatus</i>	Copepoda	43.64	0.0602
Autumn	6	<i>Calanus sinicus</i>	Copepoda	44.75	0.0583
	7	<i>Euchaeta concinna</i>	Copepoda	19.66	0.0241
	1	<i>Paracalanus aculeatus</i>	Copepoda	77.93	0.1870
	2	<i>Paracalanus</i> larva	Planktonic larvae	42.88	0.0716
	3	<i>Macrura</i> larva	Planktonic larvae	19.83	0.0434
	4	<i>Acartia pacifica</i>	Copepoda	17.24	0.0378
	5	<i>Centropages dorsispinatus</i>	Copepoda	17.68	0.0332

*Units of population density = ind.m⁻³. The value for population density was the average for the study area.

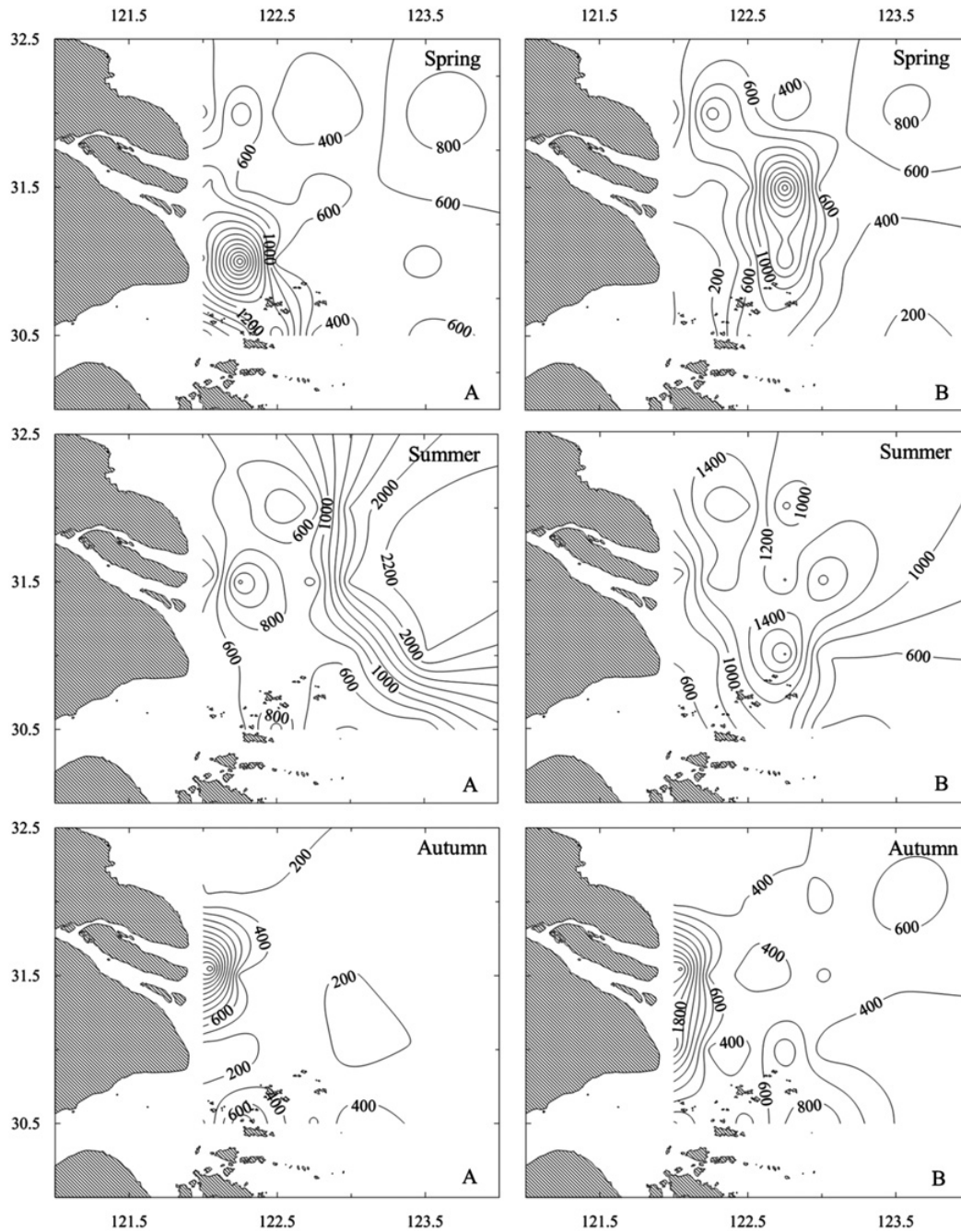


Fig. 4. Spatial patterns of population density and biomass of zooplankton in the Yangtze estuary in spring 2011, and summer and autumn 2010 (A: population density, ind.m^{-3} ; B: biomass, mg.m^{-3}).

dominant species in summer and autumn, while *Noctiluca scintillans* was the common dominant species in summer and spring. The spring and autumn shared no common dominant species except *Macrura* larva.

Distribution and seasonality of zooplankton population density and biomass

The average population density of zooplankton in the study area was 783.5, 691.3 and 399.5 ind.m^{-3} in summer,

spring and autumn, respectively, with an average population density of 624.8 ind.m^{-3} in all three seasons. In spring, the population density ranged from 71 to 3185.4 ind.m^{-3} among all sites, and decreased from inside to outside of the estuary (Fig. 4). Outside of the estuary, the population density was higher in the east than in the west. In summer, the population density ranged from 67 to 2187.5 ind.m^{-3} , and increased from inside to outside of the estuary. In autumn, the population density ranged from 33.6 to 2717.4 ind.m^{-3} . The mouth of the estuary was the only area having a population density over

Table 3. Relationships between zooplankton density and environmental variables.

Season	Determinant factors	Correlation coefficient
Spring	Temp (m)/Salinity (b)/SiO ₃ - Si (b)	0.654
	Temp (m)/Salinity (m)/SiO ₃ - Si (b)	0.646
	Temp (b)/Salinity (b)/SiO ₃ - Si (b)	0.645
Summer	Salinity (m)/SiO ₃ - Si (m)/TDN(m)	0.746
	Salinity (m)/SiO ₃ - Si (b)/TDN(b)	0.744
	SiO ₃ - Si (m)	0.744
Autumn	Temp (b)/TSS (s)/SiO ₃ - Si (m)	0.833
	Temp (b)/TSS (s)/TDN (s)	0.828
	Temp (b)/TSS (s)/TDN (m)	0.828

s, surface layer; m, middle layer; b, bottom layer; TDN, total dissolved nitrogen.

1000 ind.m⁻³ (Fig. 4). In spring, *Calanus sinicus* and *Noctiluca scintillans* were the largest two contributors to the zooplankton population density, accounting for 21.7 and 15.8%, respectively. *Acartia pacifica* and *Paracalanus aculeatus* were the main species contributing over 10% to zooplankton population density in summer and autumn (Table 2).

The average biomass in the study area varied among seasons, with 571.5, 970.6 and 613.8 mg.m⁻³ in spring, summer and autumn, respectively. In spring, the biomass ranged from 10 to 2563.8 mg.m⁻³, and the highest biomass appeared outside of the estuary (Fig. 4). In summer, the biomass ranged between 393.7 and 2021.4 mg.m⁻³, and the highest biomass also appeared outside of the estuary. The biomass ranged between 164.5 and 2437.5 mg.m⁻³ in autumn, and the highest biomass was located near the mouth of the estuary.

Relationships between zooplankton and hydro-environmental factors

BIOENV (Table 3) showed that the combination of middle layer water temperature, bottom layer salinity and bottom layer silica could explain the highest percentage (65.4%) of zooplankton assemblage variation in spring. In summer, zooplankton assemblages were most highly associated with middle layer salinity, silica and total dissolved nitrogen, with a maximum correlation coefficient of 0.746. In autumn, the combination of bottom layer water temperature, surface layer TSS and middle layer silica could explain 83.3% of zooplankton variations.

Based on these results, it appears that bottom layer salinity and temperature were essential to zooplankton assemblages for the whole year. Figure 5 shows the spatial patterns of bottom layer salinity and temperature. Bottom layer salinity exhibited a trend of increasing from the inside to the outside of the estuary during three seasons. Bottom layer temperature decreased from near-shore to offshore, with a fluctuation of 15.6–19.4 °C in spring and 21.3–30 °C in summer. In autumn, bottom layer temperatures were higher in the offshore area than the near-shore areas.

Annual zooplankton variations

The MK trend test indicated that the zooplankton biomass ($Z = 1.36$) and population density ($Z = 1.56$) have continued to increase over the past 30 years (Figs. 6 and 7). However, the composition of zooplankton species changed with time. The Medusa, Cladocera, Tunicate, Polychaeta and Decapoda population densities increased (Z -value > 0), while the population densities of the other zooplankton species decreased (Z -value < 0) over the past 30 years. However, the observed trends were all non-significant, below the confidence limit of 5 and 10% (Fig. 7).

Discussion

Spatial zooplankton distribution

The spatial distribution of zooplankton biomass was not consistent with that of zooplankton population density in spring. Maximum population density appeared at the mouth of the estuary, while maximum biomass appeared outside of the estuary. *Noctiluca scintillans* contributed strongly to total zooplankton population density, partially because in spring the increase of nutrients benefited their aggregation and partially because jellyfish assembling near the shore resulted in the decline of copepods and other competitors, mitigating the stressors of *Noctiluca scintillans*. *Calanus sinicus* contributed the most to zooplankton biomass, because they had relatively large individual weights. These two aspects together led to the inconsistent distribution of biomass and population density in the YRE. In summer, the population density and the biomass showed the same spatial patterns, such that both declined from inside to outside of the estuary due to the increase of salinity. This could also be verified from the results of BIOENV (Table 3). In autumn, water temperatures started to decrease and possibly had a greater impact on zooplankton assemblage. *Paracalanus aculeatus* constituted the most important component of zooplankton, which was adapted to the zones with both low temperature and low salinity (Xu *et al.*, 1995a). Maximum population density of zooplankton appeared inside the estuary, due to the temperature and salinity gradients, which could also be verified from the results of BIOENV (Table 3).

Seasonal zooplankton variations

Over the last decade, a significant increase in jellyfish blooms has been observed worldwide in marine ecosystems, especially in spring (Dong, 2010; Xu *et al.*, 2013). In this study, jellyfish accounted for a high proportion (13.4%) of the total population density of zooplankton in spring, while the proportion of copepods was the lowest. This phenomenon could be explained by the fact that jellyfish fed on almost all available zooplankton,

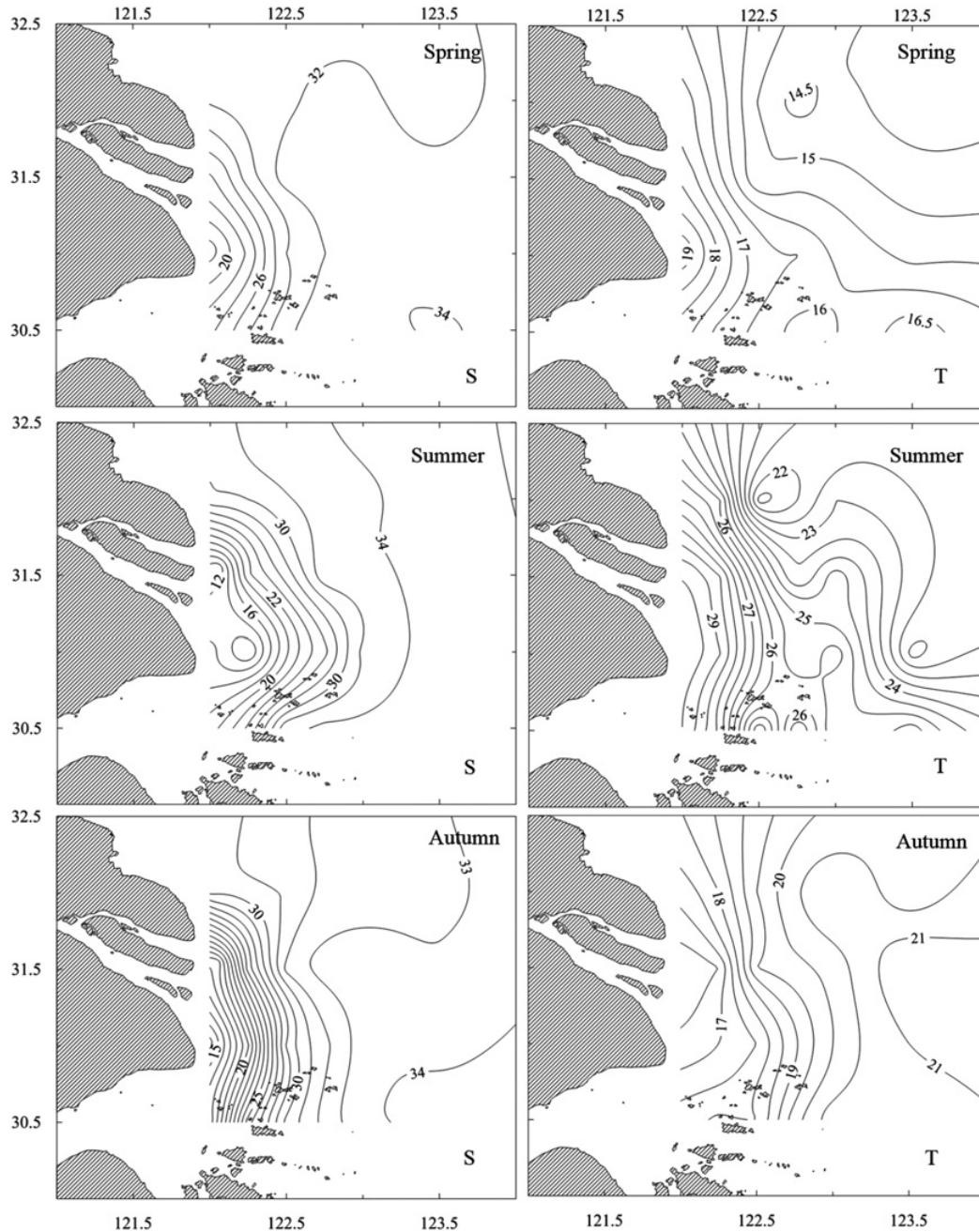


Fig. 5. Spatial patterns of bottom layer salinity and temperature in the Yangtze estuary in spring 2011, and summer and autumn 2010 (S: salinity, ‰; T: temperature, °C).

especially copepods, which controlled the copepod population. Furthermore, hydrologic regime changes and intensive human activities were all contributors to jellyfish blooms (Purcell *et al.*, 2007; Uye, 2008; Richardson *et al.*, 2009). According to the long-term oceanographic records of Chinese coastal waters (Ocean Environmental Information, 2011), it was found that the surface sea temperature rose gradually, especially in the East China Sea, where the temperature increased most prominently. The temperature rise was conducive to the growth of jellyfish. After the impoundment of the TGR, the decrease of discharge in the flood season and increase of discharge

in the dry season (Fig. 2) could lead to a favorable environment in the Yangtze estuary benefiting the growth of jellyfish. In addition, persistent riverine nutrient loading led to nutrient replenishment and consequently a dramatic rise of phytoplankton biomass (Jiang *et al.*, 2014). Phytoplankton biomass provided a rich bait foundation for the growth of zooplankton.

Dominant zooplankton species determine the path of material circulation and energy transference in a community, and also control the development of the community (Wang *et al.*, 2014). If a dominant species dies out, not only will the structure of the community change but also

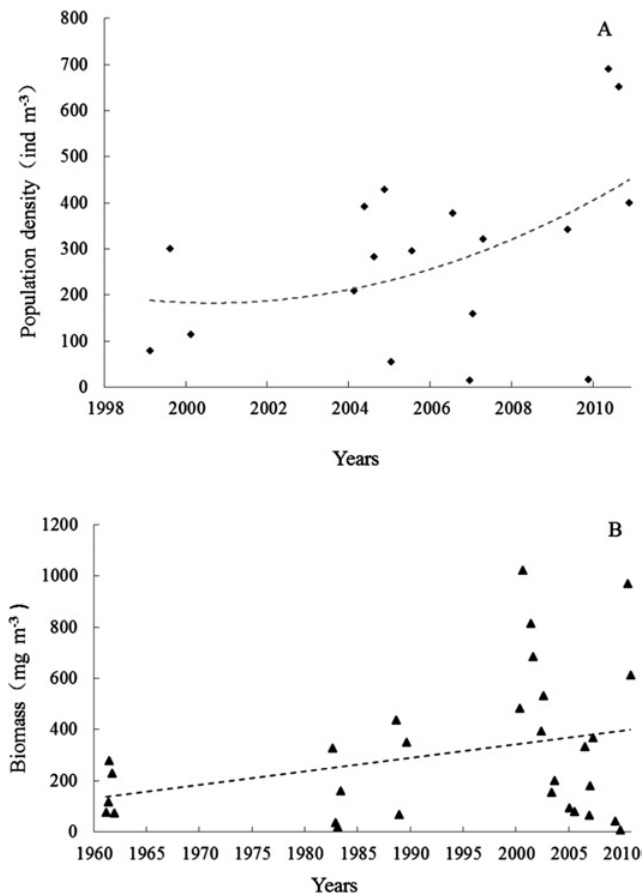


Fig. 6. Population density and biomass of zooplankton in the Yangtze River Estuary in summer over the past 30 years. Population density (ind. m^{-3}) and biomass (mg. m^{-3}) are the average values for the investigated areas.

the ecological system (Ikauniece, 2001). The dominant species of zooplankton in the study area varied among different seasons. *Noctiluca scintillans* and *Calanus sinicus* were the absolutely dominant species in spring. *Noctiluca scintillans*, whose distribution was highly related to nutrient concentration, was one of the main species of red tide in subtropical and tropical seas, and its dominance in the YRE could be caused by enhanced eutrophication (Turkoglu, 2013; Mikaelyan *et al.*, 2014). *Calanus sinicus* was the most important bait creature in the YRE, which preyed mainly on diatoms and *Coscinodiscus* sp., and supported the regional fisheries. The total abundance of *Calanus sinicus* peaked in spring, and was limited by water temperature (Xu *et al.*, 2003). *Acartia pacifica* and *Paracalanus aculeatus* were the absolutely dominant species in summer and autumn, respectively. They are both common species along the coast of China and coexist in the estuary all year round (Liang and Uye, 1996).

Long-term zooplankton variations

It could be seen that both the population density and biomass of zooplankton showed a slight but

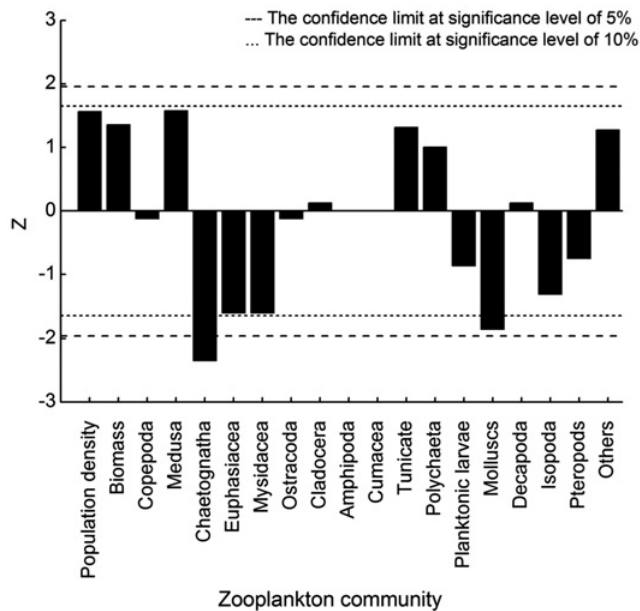


Fig. 7. Result of trend analysis of zooplankton in the Yangtze River Estuary (from 1982 to 2011). The test statistic *Z* indicates the presence of a trend; a positive (negative) value of *Z* indicates an upward (downward) trend.

non-significant ($P < 10\%$) upward trend. There are several reasons for this. Firstly, the sea surface temperature increased gradually (Belkin, 2009); in particular the increase in the East China Sea was very prominent (Jiang *et al.*, 2014). Since most species of zooplankton in the Yangtze estuary were warm-water species (Ji and Ye, 2006; Zhang *et al.*, 2009), the rise in temperatures was beneficial to the growth of zooplankton. Secondly, from a bottom-up control point of view, although the construction of dams could cause a considerable reduction in nutrient loads due to the interception of these nutrients by reservoir sedimentation, this reduction could be over-compensated by anthropogenic inputs downstream, as reported by Humborg *et al.* (1997). In addition, reservoirs have much higher interception of phosphorus (P) and silica (Si) than nitrogen (N). After TGR impoundment, nitrogen and phosphorus concentrations and in particular the N/P and N/Si ratios increased sharply (Dai *et al.*, 2011) despite deposition in the Three Gorges Reservoir. These changes induced a dramatic increase in algal biomass in the YRE (Jiang *et al.*, 2014), providing a rich bait foundation for growth and breeding of zooplankton. Finally, from the top-down control point of view, the decrease of fishery resources due to overfishing in the YRE (Chen *et al.*, 2015) alleviated the predation pressure on zooplankton. These factors together promoted the increase of population and biomass of zooplankton in the YRE.

In terms of the composition of zooplankton, the following changes were observed. Firstly, zooplanktivorous gelatinous taxa (Medusas and Tunicates) increased remarkably in the MK test. Unlike the natural phenomena of jellyfish aggregation, jellyfish blooms in pelagic ecosystems are regarded as a response to anthropogenic

disturbance and climate change (Stoner *et al.*, 2011), causing numerous deleterious consequences, such as reducing fishery production and clogging coastal power plant cooling water intakes. Secondly, the increase in the proportions of Polychaeta could be attributed to the rise in water temperatures and nutrient concentrations (Tsutsumi, 1987). Finally, Euphausiacea, Mysidacea and Mollusca exhibited a downward trend, which was possibly because they are all edible and principal fishing targets (Omori, 1978; He *et al.*, 2011).

Conclusions

- (1) The key factors affecting the zooplankton community in the YRE were salinity and temperature, implying that TGR impoundment might impact zooplankton community structure and the spatial distribution of population density and biomass through the alterations of flows, which directly affect the salinity and temperature profiles of the estuary, especially in spring and autumn.
- (2) The biomass and population density of zooplankton showed an increasing trend, and the community structure of zooplankton has also shifted over the past 30 years. The reasons for long-term changes in zooplankton were related to both TGR impoundment and nutrient inputs from the region downstream of the TGR dam.

Acknowledgements. The authors are grateful for the funding from the National Natural Science Foundation of China (No. 51425902, 91547206). Special thanks go to Dr Sun X. and Xin M. for assistance in the fieldwork and laboratory analyses. We also thank Dr Catherine Rice from the United States for proofreading the English.

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Appendix

Table A1. Zooplankton species found in the Yangtze River Estuary and adjacent sea areas in 2010–2011.

Species	Species	Species
Cladocera	<i>Acartia pacifica</i>	<i>Oithona similis</i>
<i>Penilia avirostris</i>	<i>Centropages orsinii</i>	<i>Paracalanus</i> sp.
<i>Evadne tergestina</i>	<i>Pachysoma punctatum</i>	<i>Corycaeus dahl</i>
Protozoa	<i>Oncaea clevei</i>	<i>Euchaeta plana</i>
<i>Noctiluca scintillans</i>	<i>Chiridius poppei</i>	<i>Undinula vulgaris</i>
Pteropods	<i>Candacia bradyi</i>	<i>Copilia mirabilis</i>
<i>Limacina trochiformis</i>	<i>Pontella chierchiaie</i>	<i>Tortanus forcipatus</i>
<i>Creseis virgula</i>	<i>Tortanus vermiculus</i>	<i>Paracalanus crassirostris</i>
Medusa	<i>Pleuromamma robusta</i>	<i>Eucalanus crassus</i>
<i>Aequorea australis</i>	<i>Undinula darwini</i>	<i>Setella gracilis</i>
<i>Phialidium hemisphaericum</i>	<i>Scolecithrix danae</i>	<i>Labidocera bipinnata</i>
<i>Abylopsis tetragona</i>	<i>Oncaea mediterranea</i>	<i>Acartia bifilosa</i>
<i>Beroe</i> sp.	<i>Oithona brevicornis</i>	<i>Lubbockia</i> sp.
<i>Eirene</i> sp.	<i>Pontellopsis yamadai</i>	<i>Corycaeus pacificus</i>
<i>Solmundella bitentaculata</i>	<i>Clausocalanus arcuicornis</i>	<i>Corycaeus gibbulus</i>
<i>Phialucium</i> sp.	<i>Pseudodiaptomus marinus</i>	<i>Acrocalanus gibber</i>
<i>Leusia subtiloides</i>	<i>Sapphirina nigromaculata</i>	<i>Calanopia elliptica</i>
<i>Pleurobrachia globosa</i>	<i>Corycaeus erythraeus</i>	<i>Canthocalanus pauper</i>
<i>Phialidium chengshanense</i>	<i>Sapphirina scarlata</i>	<i>Corycaeus crassiusculus</i>
<i>Obelia</i> spp.	<i>Sinocalanus sinensis</i>	<i>Acrocalanus gracilis</i>
<i>Diphyes chamissonis</i>	<i>Lucicutia flavicornis</i>	<i>Hyperioidea sibaginis</i>
<i>Liriope tetraphylla</i>	<i>Schmackeria poplesia</i>	<i>Eucalanus attenuatus</i>
<i>Muggiaea atlantica</i>	<i>Labidocera acuta</i>	<i>Labidocera minuta</i>
<i>Eirene ceylonensis</i>	<i>Euterpe acutifrons</i>	<i>Scolecithricella</i> sp.
<i>Ctenophora</i> sp.	<i>Oithona rigida</i>	<i>Microsetella norvegica</i>
<i>Aequorea conica</i>	<i>Rhincalanus cornutus</i>	<i>Paracalanus parvus</i>
Decapoda	<i>Oncaea conifera</i>	<i>Corycaeus lubbocki</i>
<i>Leptocheila gracilis</i>	<i>Tortanus derjugini</i>	<i>Oithona nana</i>
<i>Acetes chinensis</i>	<i>Corycaeus affinis</i>	<i>Nannocalanus minor</i>
Molluscs	<i>Euchaeta concinna</i>	<i>Eucalanus subcrassus</i>
<i>Creseis acicula</i>	<i>Oncaea venusta</i>	<i>Temora discadata</i>
Naticidae sp.	<i>Corycaeus andrewsi</i>	<i>Clytemnestra rostrata</i>
<i>Sepiolo birostrata</i>	<i>Corycaeus catus</i>	<i>Corycaeus</i> spp.
<i>Pterygoteuthis giardi</i>	<i>Oncaea</i> sp.	<i>Calocalanus plumulosus</i>
Copepoda	<i>Lucicutia ovalis</i>	<i>Oithona plumifera</i>
<i>Xanthocalanus multispinus</i>	<i>Sapphirina opalina</i>	<i>Scolecithrix nicobarica</i>
<i>Eucalanus mucronatus</i>	<i>Corycaeus speciosus</i>	<i>Scolecithricella longispinosa</i>
<i>Calanopia</i> sp.	<i>Acanthomysis fujingai</i>	<i>Nereidae</i> sp. Larva
<i>Paracalanus aculeatus</i>	<i>Acanthomysis longirostris</i>	<i>Macrura</i> larva
<i>Labidocera euchaeta</i>	Ostracoda	<i>Brachyura</i> larva
<i>Centropages sinensis</i>	<i>Conchoecia</i> sp.	Polychaeta
<i>Calanus sinicus</i>	<i>Euconchoecia maimai</i>	<i>Travisiopsis leviseni</i>
<i>Oncaea media</i>	<i>Zeugophilomedes polae</i>	<i>Spionidae</i> sp.
<i>Temora stylifera</i>	<i>Alacia alata major</i>	<i>Travisiopsis dubia</i>
<i>Temora turbinata</i>	<i>Euconchoecia elongata</i>	<i>Sagittella kowalewskii</i>
<i>Centropages dorsispinatus</i>	<i>Euconchoecia aculeata</i>	<i>Alciopina parasitica</i>
<i>Centropages furcatus</i>	<i>Euconchoecia</i> sp.	Amphipoda
Chaetognatha	<i>Cypridina dentata</i>	<i>Monoculodes</i> sp.
<i>Sagitta enflata</i>	Planktonic larva	<i>Lestrigonus bengalensis</i>
<i>Sagitta pulchra</i>	<i>Paracalanus</i> larva	<i>Themisto gracilipes</i>
<i>Sagitta ferox</i>	Polychaeta larva	<i>Gammaridae</i> sp.
<i>Sagitta sinica</i>	<i>Ctenophora</i> sp. larva	<i>Lysianassidae</i> sp.
Euphausiacea	<i>Acartia</i> sp. Larva	<i>Hyperioidea</i> sp.
<i>Euphausia pacifica</i>	<i>Diogenes</i> sp. larva	<i>Ampeliscidae</i> sp.
<i>Pseudeuphausia sinica</i>	<i>Syllidae</i> sp. larva	<i>Brachyscelus</i> sp.
Cumacea	Gastropoda larva	<i>Simorhynchotus antennarius</i>
<i>Lamprops hexaspina</i>	<i>Cirripedia</i> larva	<i>Lestrigonus schizogeneios</i>
<i>Bodotria ovalis</i>	Atlantidae larva	Isopoda
<i>Diastylis tricineta</i>	<i>Sthenolepis</i> sp. larva	<i>Synidotea laevidorsalis</i>
<i>Iphinoe tenera</i>	Ophiuroidea larva	Tunicate
<i>Dimorphostylis asiatica</i>	<i>Coeenterata</i> spp. larva	<i>Salpidae</i> sp.
Mysidacea	Decapoda Larva	<i>Doliolletta gegenbauri</i>
<i>Acanthomysis</i> sp.	<i>Oratosquilla</i> sp. Larva	<i>Thalia democratica</i>
<i>Acanthomysis brevirostris</i>	Fish eggs	<i>Doliolum denticulatum</i>
<i>Iiella pelagicus</i>	<i>Bivalvia</i> sp. Larva	<i>Oikopleura dioica</i>