



Three decades of sea water abstraction by Kapar power plant (Malaysia): What impacts on tropical zooplankton community?



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ABSTRACT

Zooplankton samples collected before (1985–86) and after (2013–14) the establishment of Kapar power station (KPS) were examined to test the hypothesis that increased sea surface temperature (SST) and other water quality changes have altered the zooplankton community structure. Elevated SST and reduced pH were detected between before and after impact pairs, with the greatest impact at the station closest to KPS. Present PAHs and heavy metal concentrations are unlikely causal factors. Water parameter changes did not affect diversity but community structure of the zooplankton. Tolerant small crustaceans, salps and larvaceans likely benefited from elevated temperature, reduced pH and shift to a more significant microbial loop exacerbated by eutrophication, while large crustaceans were more vulnerable to such changes. It is predicted that any further rise in SST will remove more large-bodied crustacean zooplankton, the preferred food for fish larvae and other meroplankton, with grave consequences to fishery production.

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1. Introduction

Power plants that abstract seawater to cool down its superheated power turbines generating electricity have the reputation of not only directly causing massive mortality of marine organisms by impingement and entrainment (Lewis and Seegert, 2000; Greenwood, 2008; Azila and Chong, 2010), but also indirectly killing or undermining organisms via thermal discharges that increase temperature, acidification and chemical pollution of the receiving waters (Bamber and Spencer, 1984; Poornima et al., 2005; Ferry-Graham et al., 2008; Chuang et al., 2009; Coulter et al., 2014). A single once-through cooled (OTC) power plant can withdraw millions of cubic meters of cooling-water daily before discharging the heated water including chemical biocides into the sea. Impingement mortality at the intake screens is highly variable, ranging from annual fish deaths of about 13,000 in a 930 MW power plant in USA to 214 million in a 5706 MW power plant in France (see Azila and Chong, 2010). Small nekton and plankton are however subject to entrainment into the power plant's heat-cooling system. The annual entrainment mortality of larval fish is usually very high ranging from 132 million fish eggs and larvae (or 23% of the total ichthyoplankton) through a single small direct-cooled power plant such as at the Wabash River, USA (Lewis and Seegert, 2000) to an estimated 10^{14} fish larvae in north European waters where 45 large power plants were located (Henderson, 2015).

Discharged thermal waters are 2–12 °C warmer than intake temperatures in temperate regions (Bamber, 1995; Hoffmeyer et al., 2005; Coulter et al., 2014), 8–12 °C in a subtropical region (Chuang et al., 2009), while differences of 7–10 °C are reported in tropical regions (Anton, 1990; Poornima et al., 2005; Pokale, 2012). Phytoplankton biomass especially during spring was significantly greater in the intake region than in the outlet region of a nuclear power plant in subtropical Taiwan (Chuang et al., 2009). Warmed water seems to limit the growth of more nutritious and palatable large diatom species, but favors the less palatable and small diatom species, cyanobacteria and harmful algae (Hutchins et al., 2007). Mortality of copepods, cirriped and crab larvae was higher at the water discharge outlets compared to intake points (Hoffmeyer et al., 2005; Choi et al., 2012), with increased thermal sensitivity in the presence of chlorine (Choi et al., 2012). Warm water also induces high productivity of heterotrophic bacteria in the microbial loop of marine food webs (Berglund et al., 2007; Sarmiento et al., 2010), which may alter the zooplankton community in such a way that species (for instance, large-sized copepods) that rely on large nutritious diatoms are eliminated from the system, whereas species (small-sized ones) that rely on heterotrophic microbes and small plankton become dominant in the system (Richardson, 2008). Richardson (2008) concluded that the replacement of large-sized copepods by small-sized ones (as well as the blooming of jellyfish and ctenophores) is the likely consequence of sea warming. Striking phenological changes due to sea warming have been documented in temperate waters including for dinoflagellates, diatoms, holozooplankton and meroplanktonic larvae where their seasonal peaks advanced by 23 days, 22 days, 10 days and

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27 days, respectively, indicating asynchronous responses to ocean warming that could lead to trophic mismatches in the marine food web (Edwards and Richardson, 2004).

Power plants also discharge thermal water that contains biocidal chemicals, especially chlorine that is widely used to control biofouling organisms in the cooling systems. Chlorination has a suppressive effect on phytoplankton (Chuang et al., 2009) although laboratory tests on *Skeletonema costatum* and *Chlamydomonas* sp. showed no damage caused by high temperatures (35–40 °C) and residual chlorine (Hirayama and Hirano, 1970). The formation of chlorination byproducts may have potential harmful effects on biota (Choi et al., 2012). Power plants also generate a variety of solid wastes. Coal combustion creates ash piled into ash ponds where ash leach out of boron, arsenic, lead, mercury and polycyclic aromatic hydrocarbons (PAHs) by rainwater may leak out to contaminate groundwater and seawater. The coal-fired power plant's flue gas stacks emit water vapors, fine particulate matter, carbon dioxide and various air pollutants including carbon monoxide, nitrogen oxides (NO_x), sulfur dioxide (SO₂), copper and mercury which could be precipitated as acid rain acidifying and polluting the environment if uncontrolled (Shindell and Faluvegi, 2010; Guttikunda and Jawahar, 2014).

There are currently seven thermal power plants in Malaysia, four of which are OTCs located along the Straits of Malacca. The environmental impacts of the siting and operation of these power plants in the country are largely unknown. In the only impingement study at the Kapar power plant in the country, Azila and Chong (2010) reported an estimated 8.5×10^6 (42 tons) marine organisms succumbed annually due to impingement, the bulk (178 species) of which comprised shrimp (64.5%) and fish (25.9%). To date, there is however no entrainment studies done at any of the seven power stations. Neither is there any work done on other environmental impacts of these power stations on receiving waters, as for example the lethal and chronic effects caused by the discharged thermal and chemical effluents.

Kapar power station (3°7'1"N, 101°19'1"E) or the Sultan Salahuddin Power Plant is located at the Kapar estuary, 24 km north of Port Klang. It was established in 1987 to cater for the power demand of rapid industrial development and human population growth in the hinterland which includes the city of Kuala Lumpur. The Kapar power station (KPS) is the largest power plant in Malaysia, generating 2420 MW of electricity using coal-, oil- and gas-fired thermal energy. It is an OTC power station withdrawing seawater at the rate of about $70 \text{ m}^3 \text{ s}^{-1}$ from the Klang Strait. There are six intake points for turbine cooling and the heated cooling water is discharged back into the strait at a combined rate of up to 6 million m^3 daily (Azila and Chong, 2010). Thermal plumes from the power plant during high tide could yield temperatures of up to 36 °C (Anton, 1990). Zooplankton samples and environmental data collected in July 1985 to October 1986 from the Klang Strait just before the operation of KPS are available in our archived collections. This offers an opportunity to investigate the impact of KPS on the surrounding waters, in particular, the zooplankton community in Klang Strait. Since zooplankton organisms are ectotherms with short generation times, the use of zooplankton as sentinels to monitor the effects of climate change on marine biota may be useful (Dam, 2013). Moreover, zooplankton forms the crucial links between phytoplankton and higher trophic levels, and also constitutes the major food base that supports fisheries. Nevertheless, understanding the effects of the power plant on the zooplankton is challenging given the multiple anthropogenic stressors and varied biotic responses.

Here, we hypothesize that entrainment and sea temperature rise due to KPS with water quality changes due to nearby anthropogenic activities have altered the zooplankton community. We used an asymmetrical 'Before-After-Control-Impact' or Beyond BACI experimental design to assess the assumed impact due to KPS, following Lardicci et al. (1999) in a similar assessment of power plant impact on benthic communities. The design consisted of one putatively impacted and several control sites to distinguish between environmental impacts/changes due to

impact and natural background variability (Underwood and Chapman, 1997). The objectives of the study were to compare the zooplankton community structure before and after impact, determine the structural changes resulting from the impact, and discuss the implications of such changes. Although this study assumes a single anthropogenic perturbation (KPS), we also consider the possible co-effects of PAH and heavy metal pollution in the study site due to port activities south of KPS.

2. Materials and methods

2.1. Study site

The study site was located in Klang Strait situated on the southwest region of Peninsular Malaysia (Fig. 1). Klang Strait stretches 70 km northwestwards from its southernmost entrance located south of several deltaic islands to north of Angsa Bank where its northern entrance funnels out into the Straits of Malacca. Angsa Bank is a huge, tongue-like sandbank that extends 40 km northwestward from Klang Island, nourished by continuous sediment transport from the south. The main coastal vegetation in the area is mangrove partly cleared on the mainland leaving the largest tract (ca. 15,000 ha) of remaining mangrove forests of the state of Selangor on the deltaic islands. Selangor produces approximately 120,000 tonnes of marine fish annually making it the second largest marine fish producing state of Malaysia. The depth at Klang Strait's northwest entrance exceeds 18 m allowing access to large ships but only small tug boats to enter farther into the country's largest port, Port Klang. The freshwater inflows are mainly from the rivers of Langat and Klang in the south and Selangor River in the north. The source of freshwater inputs is attributable to heavy rainfalls dictated by the northeast (NE) monsoon (November to March) and southwest (SW) monsoon (May to September). The inter-monsoon transition period (April and October) normally brings heavy rain. The tidal regime is semidiurnal and tidal heights at Mean High Water Springs (MHWS), Mean Low Water Springs (MLWS), Mean High Water Neaps (MHWN) and Mean Low Water Neaps (MLWN) averaged 5.2 m, 1.0 m, 3.9 m and 2.5 m above chart datum, respectively (Tide Tables Malaysia, Hydrographic Directorate, Royal Malaysian Navy). Tides are macrotidal with strong tidal velocities reaching 1.5 m s^{-1} during spring tide and 0.4 m s^{-1} during neap tide (Chong et al., 1996).

2.2. Field methods

The asymmetrical BACI sampling design (Underwood, 1992) was used to investigate the abundance and community structure of zooplankton before and after power plant impact in the Klang Strait. Two sets of zooplankton samples were compared; one set collected before KPS operation (thereafter referred to as before-impact) from July 1985 to June 1986 (archived collections of Chong), and another set collected almost 30 years thereafter from August 2013 to April 2014 (after-impact). Sampling was conducted monthly before and after impact except in January 2014. The after-impact zooplankton collections were conducted at five stations along an 18-km transect in the Klang Strait, exactly replicating the five stations established before-impact. The assumed most impacted station (I) was located 45 m from KPS, while the other four stations (C1 to C4) constituting the controls (Cs) were located progressively away from KPS. Interval distance between any two stations was 4.5 km and the shoreline distance of C1 to C4 ranged from 1.8 km to 10.5 km. The mean water depths along the transect from Stn I through C1, C2, C3 to C4 were 15.1 m, 10.8 m, 14.0 m, 14.4 m and 22.5 m, respectively.

At each station, water parameters like temperature, salinity and pH were measured at 0.5 to 1 m from the surface before and after impact. The before-impact conditions were recorded by a transparent 3-L Van Dorn-type water sampler with an internal mercury-column thermometer (KAHLISCO 135WC203), salinity–conductivity–temperature meter (YSI 33, Yellow Springs, USA) and a digital pH-temperature meter

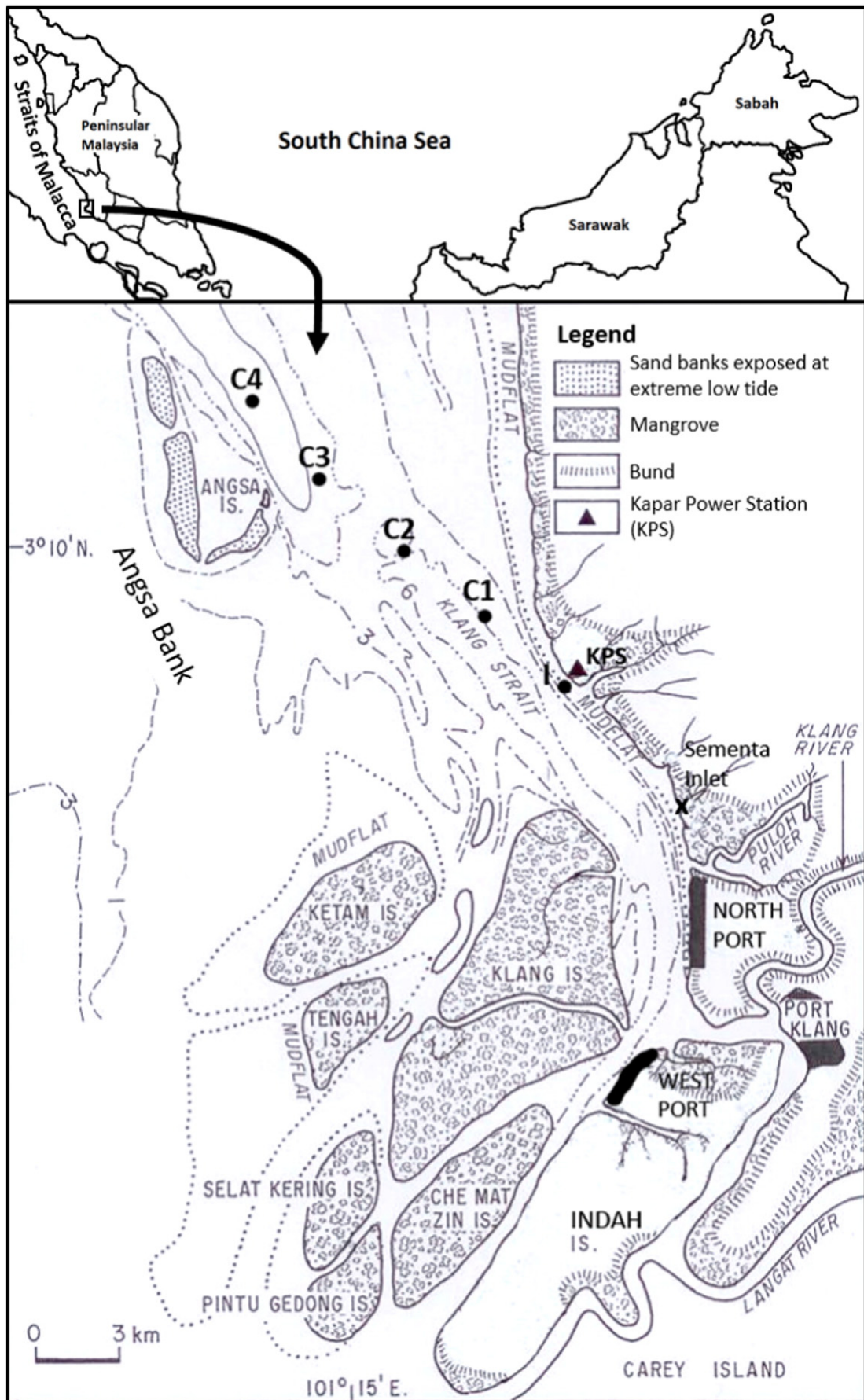


Fig. 1. Location of sampling stations (filled circles) along established 18 km transect in Klang Strait, Peninsular Malaysia.

(ORION Research 221, USA). The after-impact conditions were recorded using calibrated multi-parameter probe (YSI 556 or Hydrolab 4b, USA). In addition, sea surface temperature (SST) was continuously recorded after impact over 24 h using deployed HOBO pendant temperature loggers (UA-002-64, Onset Computer Corporation, USA) placed at Stn I and Stn C4 on two occasions during the dry and wet period. Analyses of PAHs, dissolved inorganic nutrients and chlorophyll *a* concentration from water samples taken at each station were only done after impact. The sampled water was immediately stored in glass bottles kept cold in an ice-chest. Zooplankton samples were collected by the same 45 cm-diameter MARMAP bongo nets (KAHLSICO 013WA165, USA) fitted with calibrated flow-meters and twin nets of 363 and 180 μm mesh sizes before and after impact. At least two oblique tows from 1 m above the seabed to the surface were made at each station. The total volume of net filtration ranged from 7.5 m^3 to 40.5 m^3 depending on the water depth. Zooplankton samples were preserved with 4% borax-buffered neutralized formaldehyde solution in seawater for subsequent laboratory analysis.

2.3. Laboratory procedures

Chlorophyll pigments of phytoplankton in 100 mL of sampled seawater were extracted by adding 10 mL of 90% acetone into a 10 mL polypropylene tube containing the sieved phytoplankton on its filter paper (Whatman GF/C). Chlorophyll extraction was completed after 16 h in a freezer at -20°C . The fluorescence of extracted chlorophyll before and after acidification (10% HCl) was measured by a 10AU Turner Design fluorometer. Chl. *a* concentration was then calculated based on the equations given in Parsons et al. (1984). Dissolved inorganic nutrients (DIN) comprising of $\text{NH}_3\text{-N}$, $\text{NO}_2\text{-N}$, $\text{NO}_3\text{-N}$ and PO_4^{3-} were measured by HACH DR/2010 spectrophotometer based on the methods outlined in HACH Water Analysis Handbook (HACH, 1997). Ten types of PAHs were analyzed, namely, naphthalene, acenaphthylene, acenaphthene, fluorene, phenanthrene, anthracene, fluoranthene, pyrene, benzo[a]anthracene and chrysene. The PAH extraction was performed using the solid-phase extraction (SPE) procedure adopted by Kraleva et al. (2012). The extracted PAH components were analyzed by gas chromatographic method (Shimadzu GC-17A, Japan). Calibration curves were obtained using a series of varying concentrations of a multicomponent standard containing each of the ten PAHs. The concentrations of Chl. *a*, DIN and PAHs were expressed in units of $\mu\text{g L}^{-1}$, μM and $\mu\text{g L}^{-1}$, respectively.

As the 363- μm net targeted mainly the larger zooplankton and would under-sample the small-bodied copepods, only zooplankton samples from the 180- μm mesh net were enumerated and reported here. Each zooplankton sample was gently sieved through stacked Endecott sieves with size fractions of 1000 μm , 500 μm , 250 μm and 125 μm . The entire 1000- μm fraction was enumerated if organisms were at low abundance; otherwise it was split using a Folsom plankton splitter. For smaller fractions (500 μm , 250 μm and 125 μm), zooplankton were subsampled using a 1-mL Stempel pipette and transferred onto a 1-mL Sedgewick rafter cell for enumeration under a microscope (Olympus BX50 and Leica M125). At least 200 zooplankton organisms were enumerated from each subsample. Zooplankton were identified to family level using relevant texts (Bhaud and Cazaux, 1987; Chihara and Murano, 1997; Young et al., 2002; Boxshall and Hasley, 2004). The abundance of zooplankton was estimated as individuals per cubic meter (ind m^{-3}).

2.4. Data analysis

2.4.1. Diversity measures

Taxa richness (*S*), Shannon–Wiener diversity index (*H'*) (Shannon, 1948) and Pielou's evenness (*J'*) (Pielou, 1969) were calculated for the zooplankton community which comprised 99 taxa excluding fish larvae

and penaeid larvae. The index *H'* and *J'* were used to depict the individual distribution among the zooplankton taxa. The highest values indicate that the abundance of many zooplankton taxa was equally represented or without dominance of any taxa. The diversity measures were computed by using the Plymouth Routines in Multivariate Ecological Research (PRIMER 6) software.

2.4.2. Univariate analysis

A non-parametric Kruskal–Wallis ANOVA test was applied on chlorophyll *a*, DIN and PAHs data to determine the spatial variability of these parameters. Non-parametric Spearman rank correlation was used to test the significant correlation between chlorophyll *a* and DIN. A two-way ANOVA with uneven replicates was used to detect any significant difference of the univariate data between before and after impact (BA) and among sampling stations (*S*). In order to detect correctly the zooplankton variability that was due to the effect of KPS or natural temporal change at the impacted area, the factor *S* was further partitioned into two components. The first component compared the impacted station (I) versus pooled control stations (Cs) before and after impact, and the second component compared among the Cs before and after impact. Any effect of KPS on the zooplankton community could be detected by the significant interactions of BA \times I vs. Cs. On the other hand, the ANOVA test on the component of among Cs reveals the potential variability due to spatial heterogeneity (Underwood, 1992). The dependent variables that were subjected to ANOVA tests were *H'*, *J'* and the abundance data of total zooplankton and nine key taxa (i.e. Paracalanidae, Oithonidae, Oikopleuridae, Acartiidae, Luciferidae, Euterpinae, Eucalanidae, Centropagidae and Chaetognatha). The key taxa were selected based on the magnitude of their percentage composition and occurrence. Prior to ANOVA tests, abundance data were fourth-root or $\log_{10}(x + 1)$ -transformed so as to fulfill parametric assumptions. The statistical analyses were computed using the Statistica Version 8 software.

2.4.3. Multivariate analysis

Analogous to ANOVA but involving multiple dependent variables, the multivariate technique PERMANOVA was used to detect the significant differences in zooplankton community structure over space and time. This technique was based on the permutation method with the pseudo-*F* statistic generated from the resemblance matrix data (Anderson et al., 2008). Prior to PERMANOVA, abundance of 99 zooplankton taxa was $\log_{10}(x + 1)$ -transformed to normalize the ecological data (Legendre and Legendre, 1998). The resemblance matrix among samples was then calculated based on the Bray–Curtis similarity coefficient (Bray and Curtis, 1957). The significant *p* value was eventually obtained from 9999 random permutations performed on the resemblance matrix. The PERMANOVA was computed using the PRIMER 6 and PERMANOVA+ software packages.

The principle coordinate analysis (PCO), also called metric multidimensional scaling, was used to depict the zooplankton community structure as influenced by before–after impact and control conditions. This analysis extracts the variance from the resemblance matrix (Bray–Curtis distance) and projects it onto the axes of normally a two-dimensional diagram. To apply the PCO, abundance data of the top most abundant 45 zooplankton taxa were $\log_{10}(x + 1)$ -transformed before analysis. PCO was performed using the PrCoord 1.0 in the CANOCO 4.5 software (Ter Braak and Smilauer, 2002).

3. Results

3.1. Environmental parameters

SST recorded before (1985/86) and after (2013/14) the operation of KPS ranged from 26.00 $^\circ\text{C}$ to 30.60 $^\circ\text{C}$ ($29.34 \pm 1.09^\circ\text{C}$, $n = 68$) and 28.82 $^\circ\text{C}$ to 30.98 $^\circ\text{C}$ ($29.92 \pm 0.66^\circ\text{C}$, $n = 82$), respectively (Table 1). Under after-impact conditions, SST was time-dependent on the daily

Table 1

Physical parameters recorded at before–after impacted station I and control stations (C1 to C4) along transect to 18 km offshore in Klang Strait, Peninsular Malaysia.

Environmental variable		Station									
		I		C1		C2		C3		C4	
		Before	After	Before	After	Before	After	Before	After	Before	After
Sampling time range		13.10–18.10	9.00–12.00	13.03–17.40	9.48–13.53	10.30–16.30	11.25–14.37	14.00–15.37	10.52–15.14	12.17–13.51	11.29–15.49
Temperature (°C)	Mean	29.52	30.08	29.48	29.67	29.24	29.94	29.25	29.93	29.24	29.95
	n	14	21	14	15	14	15	13	15	13	16
	sd	1.10	0.71	1.14	0.53	1.18	0.73	1.12	0.68	1.05	0.62
Salinity	Mean	30.0	29.5	30.2	30.1	30.6	30.7	30.8	31.1	31.2	31.6
	n	6	21	6	15	6	15	4	15	5	18
	sd	0.8	1.4	0.7	1.7	0.9	1.5	0.5	1.3	1.4	1.4
pH	Mean	7.95	7.47	8.06	7.68	8.06	7.72	8.05	7.77	8.08	7.80
	n	7	21	6	15	8	15	9	15	6	18
	sd	0.24	0.15	0.21	0.13	0.16	0.11	0.17	0.09	0.17	0.09
Dissolved oxygen (mg L ⁻¹)	Mean	5.1	4.8	5.5	5.2	5.4	5.6	5.1	6.1	5.4	6.3
	n	15	12	14	6	16	6	13	6	16	10
	sd	0.8	0.4	0.8	0.2	0.8	0.6	0.9	0.5	0.7	1.2

cycle of temperature monitored by HOBO pendants, with peak SST recorded in the afternoon (Fig. 2). The highest SST was recorded at Stn I after impact even though field measurements were always made in the morning (Table 1) when SST was the lowest. After pooling all control stations, mean SST before and after impact was 29.30 ± 1.10 °C and 29.87 ± 0.64 °C respectively. Diurnal variability in SST recorded over 30 h at the impacted Stn I ranged from 30.46 °C to 33.01 °C (31.45 ± 0.67 °C) during the drier SW monsoon period (May 2014) and 30.15 °C to 31.88 °C (30.63 ± 0.34 °C) during the wetter NE monsoon period (December 2014), respectively (Fig. 2a). At the control Stn C4, diurnal SST varied between 30.05 °C and 31.47 °C (30.62 ± 0.36 °C) during the dry period, and between 29.95 °C and 30.86 °C (30.31 ± 0.25 °C) during the wet period (Fig. 2b). Temperatures were generally above the daily average between 12 PM and 2 AM and below the daily average between 2 AM and 10 AM (Fig. 2).

There was no significant difference in mean salinity (<0.5) before and after impact (Table 1). A small salinity gradient was observed from Stn I to Stn C4 under both before-impact and after-impact conditions. Mean pH value among stations was lower after impact than before impact (Table 1). The overall mean pH value before impact was 8.04 ± 0.22 ($n = 36$), while the overall mean after impact was 7.67 ± 0.17 ($n =$

84). Mean pH value increased from Stn I (before/after impact: $7.95 \pm 0.24/7.47 \pm 0.15$) to Stn C4 ($8.08 \pm 0.17/7.80 \pm 0.09$) (Table 1). Mean dissolved oxygen concentration before and after impact ranged from 5.1 to 5.6 mg L⁻¹ and 4.8 to 6.3 mg L⁻¹, respectively (Table 1).

Total PAHs concentration after impact was not significantly different between I and Cs (Kruskal–Wallis ANOVA, $p > 0.05$), with mean value that ranged from 0.56 ± 1.27 µg L⁻¹ at Stn C1 to 1.80 ± 3.78 µg L⁻¹ at Stn C2 (Table 2). The concentration of dissolved inorganic nitrogen (NO_3^- -N and NH_3 -N) was significantly higher at Stn I as compared to the farthest station C4 (Kruskal–Wallis ANOVA, $p < 0.01$; Table 2). Mean dissolved inorganic phosphate (PO_4^{3-}) was not significantly different among stations (Kruskal–Wallis ANOVA, $p > 0.05$). Chlorophyll *a* concentration was higher during the SW and inter-monsoon period as compared to the NE monsoon period. An exceptional peak in chlorophyll *a* concentration was measured in October 2013 (30.2 ± 11.5 µg L⁻¹) due to *S. costatum* bloom. As indicated by Spearman rank correlation, chlorophyll *a* concentration was significantly correlated with NO_2^- -N and NO_2^- -N + NO_3^- -N ($r > -0.3$), and PO_4^{3-} concentrations ($r = -0.34$) ($p < 0.05$) but appeared not to be significantly correlated with NH_3 -N ($p > 0.05$). There was no significant difference in chlorophyll *a* concentration among all stations (Kruskal–Wallis ANOVA, $p > 0.05$; Table 2).

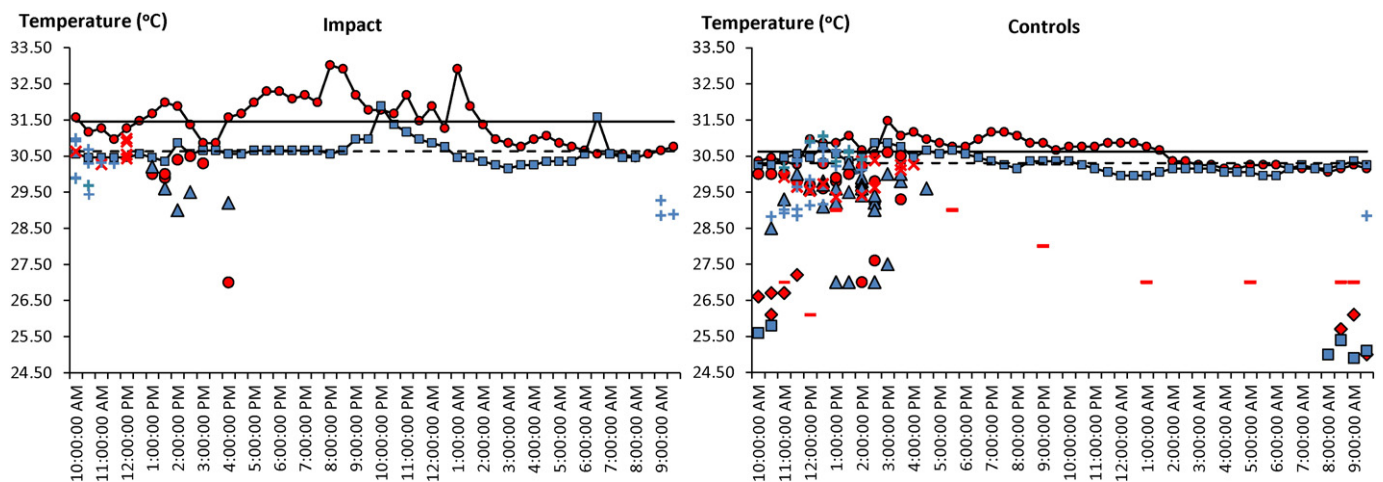


Fig. 2. Diurnal temperature fluctuations at impacted (I) and control station (C4) during the dry (circle-solid line, May 2014) and wet (square-solid line, December 2014) periods in Klang Strait. Mean daily temperature plotted as horizontal lines (solid line, May; broken line, December). Plotted free symbols indicate temperature measured before (1985/86, circle and triangle) and after impact (2013/14, cross and plus) at I and Cs stations; red symbols indicate readings during SW Monsoon (April to September), blue symbols during NE Monsoon (October to March). Additional temperature records at the mouth of Sementa Inlet (labeled X, Fig. 1) during the mid-70s are indicated by 'dash' (June to September 1975–77; Chong, 1975; Tan, 1976; Fong, 1978), 'diamond' (April to September 1976–77, Chong, 1979), and 'square' (October to March 1976–77, Chong, 1979).

Table 2Mean chlorophyll *a*, dissolved inorganic nutrients and PAHs concentrations measured from impacted station (I) and four control stations (C1–C4) along Klang Strait.

Environmental parameter	n	Station				
		I	C1	C2	C3	C4
Chlorophyll <i>a</i> ($\mu\text{g L}^{-1}$)		24 (12)	16 (12)	16 (12)	16 (12)	14 (12)
$\text{NO}_2^- - \text{N}$ (μM)		5.8 \pm 6.2	9.9 \pm 16.3	5.5 \pm 7.7	6.6 \pm 8.8	5.9 \pm 7.1
$\text{NO}_2^- - \text{N} + \text{NO}_3^- - \text{N}$ (μM)		4.96 \pm 5.29	4.95 \pm 5.84	3.94 \pm 5.5	3.04 \pm 4.18	2.66 \pm 4.33
$\text{NH}_3 - \text{N}$ (μM)		9.25 \pm 4.78	8.52 \pm 5.3	6.81 \pm 4.15	5.38 \pm 3.49	3.53 \pm 3.02
PO_4^{3-} (μM)		9.46 \pm 7.23	4.82 \pm 3.38	2.99 \pm 2.68	2.57 \pm 3.09	2.14 \pm 2.33
Total PAHs ($\mu\text{g L}^{-1}$)		2.69 \pm 4.05	1.88 \pm 1.71	1.95 \pm 2.18	2.33 \pm 4.40	1.56 \pm 1.52
		0.98 \pm 1.62	0.56 \pm 1.27	1.80 \pm 3.78	1.32 \pm 2.61	1.27 \pm 2.43

Boldface indicates significant highest value among stations (Kruskal–Wallis ANOVA, $p < 0.05$).

3.2. Zooplankton diversity

The total number of zooplankton taxa recorded before and after impact were 90 and 91 taxa, respectively. Before impact, taxa richness ranged from 69 to 76 taxa per station, and after impact it ranged from 71 to 80 taxa (Table 3). Out of the 99 taxa pooled from before and after impact samples, copepods constituted 21 taxa. Three taxa (i.e. Paracalanidae, Oithonidae and Chaetognatha) were present in all samples (100% occurrence) while 37 taxa were uncommon (<20% occurrence, Table 3). Shannon–Wiener index H' and Pielou's evenness J' of the zooplankton community were not significantly affected (ANOVA, $p > 0.05$; Table 4), spatially (i.e. among stations) and temporally (i.e. before and after impact).

3.3. Comparisons of zooplankton abundance and composition

Mean total abundance of zooplankton at Stn I was not significantly different from that of controls (ANOVA, $p > 0.05$). There was also no significant difference in mean total zooplankton abundance among control stations (ANOVA, $p > 0.05$, Table 4). However, total zooplankton abundance at impacted and control stations was significantly higher after impact ($19,599 \pm 14,018$ ind m^{-3}) than before impact (8772 ± 5675 ind m^{-3}) due to the high abundance of three dominant taxa, namely, the copepods Paracalanidae and Oithonidae and the appendicularian Oikopleuridae (ANOVA, $p < 0.01$; Table 4, Fig. 3). There were at least 69 taxa recorded at each sampling station but only a few taxa were predominant among zooplankton before and after impact. These were the copepods Paracalanidae (38.0% of overall percentage composition), Oithonidae (18.5%), Euterpinae (3.8%), Acartiidae (3.5%) and Eucalanidae (2.5%), and Oikopleuridae (7.0%), Cirripedia (4.6%) and Chaetognatha (1.7%). More than 38% of the zooplankton abundance was dominated by the calanoid copepod Paracalanidae at all sampling stations before impact (Table 3). The percentage composition of Paracalanidae after impact ranged from 31.7% at Stn C4 to 39.1% at Stn C1 (Table 3). Although percentage composition had decreased, the mean Paracalanidae abundance had increased significantly from 3923 ± 2746 ind m^{-3} before impact, to 6931 ± 7610 ind m^{-3} after impact (ANOVA, $p < 0.05$) at both Stn I and control stations (Table 4). However, the abundance of Paracalanidae was not significantly affected along the transect (ANOVA, $p > 0.05$; Table 4).

Another key copepod family, the cyclopoid Oithonidae made up <5.9% of the zooplankton abundance before impact but had increased up to 29.5% after impact (Table 3). Two species namely *Oithona attenuata* and *Oithona simplex* were predominant. ANOVA results revealed no significant difference in abundance of Oithonidae among sampling stations (ANOVA, $p > 0.05$) but its abundance after impact (4689 ± 4338 ind m^{-3}) was ten times higher than before impact (413 ± 301 ind m^{-3}) (ANOVA, $p < 0.01$; Table 4, Fig. 3a). The appendicularian Oikopleuridae was the most abundant group after Paracalanidae and Oithonidae after impact, with percentage composition that ranged from 7.4% to 11.0% (Table 3). However, the percentage composition of Oikopleuridae did not exceed 2% among sampling stations before impact (Table 3). This taxon appeared to be more common

after impact with percentage occurrence that ranged from 88% to 100% compared to 62% to 80% before impact (Table 3). A tenfold increase in abundance of Oikopleuridae (ANOVA, $p < 0.01$; Table 4) was observed after impact (1810 ± 2582 ind m^{-3}) than before impact (107 ± 184 ind m^{-3}). The distribution of Oikopleuridae was not affected by sampling location (ANOVA, $p > 0.05$; Table 4).

Cirriped nauplii were significantly more abundant at Stn I than control stations (ANOVA, $p < 0.01$) after impact but no significant difference was detected among controls (ANOVA, $p > 0.05$; Table 4). Cirripeds constituted 12.3% of zooplankton abundance at Stn I and <2% at control stations after impact (Table 3). In contrast, the calanoid copepod Centropagidae was less abundant at Stn I as compared to control stations after impact (ANOVA, $p < 0.01$; Table 4). Both Cirripedia and Centropagidae were significantly more abundant after impact than before impact (ANOVA, $p < 0.01$; Table 4). The occurrence of these two taxa was also higher after impact (75% to 100%) than before impact (67% to 94%; Table 3). The abundance of the calanoid Acartiidae and sergestid shrimp Luciferidae was significantly different among stations and between before and after impact conditions (ANOVA, $p < 0.01$; Table 4). Acartiidae was significantly more abundant at Stn I than controls before and after impact (ANOVA, $p < 0.01$; Table 4). There was also a significant difference in abundance of Acartiidae among controls (ANOVA, $p < 0.01$) suggesting spatial variability in their abundance (Table 4). Although there was no significant difference between Stn I and controls (ANOVA, $p > 0.05$), the interactions between BA and I vs. Cs indicated that the abundance of Luciferidae at Stn I was significantly lower after impact than before impact. There was also no significant difference detected for abundance of Luciferidae between Stn I and controls before impact; this however changed after impact, with significantly lower abundance at Stn I than Stn Cs (ANOVA, $p < 0.01$; Table 4). Overall, both Acartiidae and Luciferidae at Stn I and Stn Cs encountered a significant reduction in abundance after impact. The magnitude of reduction was very obvious at Stn I (ANOVA, $p < 0.01$) as compared to Stn Cs (ANOVA, $p < 0.05$; Table 4, Fig. 3b, c). Acartiidae at Stn I constituted 9.9% of zooplankton abundance before impact but comprised 2.5% after impact (Table 3). Similarly, Luciferidae (*Lucifer hanseni*) was present in all samples collected before impact (100% occurrence), but their occurrence fell especially at Stn I (83%) after impact (Table 3).

The large calanoid Eucalanidae (*Subeucalanus subcrassus*) was also likely affected by the power station as indicated by interactions between BA and I vs. Cs (ANOVA, $p < 0.01$; Table 4). The abundance of this taxon was not significantly different between Stn I and Stn Cs before impact, but had decreased significantly at Stn I after impact. Eucalanidae appeared to be more abundant at Stn Cs after impact (ANOVA, $p < 0.01$; Fig. 3e). Post-hoc Tukey–HSD test did not reveal any significant difference among all BA \times I vs. Cs pairs ($p > 0.05$) for Euterpinae (*Euterpina acutifrons*). Neither spatial nor time difference affected the abundance of Chaetognatha in the Klang Strait (ANOVA, $p > 0.05$; Table 4).

3.4. Comparisons of zooplankton community structure

PERMANOVA tests revealed statistical differences in zooplankton community structure (Bray–Curtis similarity) between impact-control

Table 3

Mean abundance (ind m⁻³), percentage composition of abundance (%) and frequency of occurrence (FO) of zooplankton over sampling stations before and after impact. <1 denotes less than 1 individual or percent, n sample size and S species richness. Only the abundance of the top 45 taxa is shown. Symbols in parentheses after taxa, '–', '0' and '+' denote vulnerable, resilient and opportunistic group, respectively.

Station	I		C1		C2		C3		C4	
Sampling period	Before	After	Before	After	Before	After	Before	After	Before	After
n	16	24	13	16	15	16	15	16	17	14
Acartiidae (–)	1013	559	903	197	686	720	306	227	253	214
% (FO)	9.9(100)	2.5(100)	8.0(92)	1.0(100)	7.4(100)	3.6(100)	4.0(100)	1.3(100)	4.1(100)	1.3(100)
Centropagidae (0)	44	92	136	230	107	443	69	259	36	179
% (FO)	<1(69)	<1(75)	1.2(85)	1.1(88)	1.2(87)	2.2(100)	<1(93)	1.5(100)	<1(88)	1.1(100)
Calanidae (–)	36	4	60	35	72	113	79	329	92	104
% (FO)	<1(81)	<1(33)	<1(85)	<1(75)	<1(100)	<1(94)	1.0(93)	1.9(94)	1.5(94)	<1(100)
Eucalanidae (–)	371	189	336	488	352	731	199	507	257	332
% (FO)	3.6(100)	<1(92)	3.0(100)	2.4(100)	3.8(100)	3.6(100)	2.6(100)	2.9(100)	4.2(100)	2.1(100)
Euchaetidae (–)	38	41	18	205	41	136	57	144	46	121
% (FO)	<1(81)	<1(38)	<1(69)	1.0(88)	<1(93)	<1(94)	<1(87)	<1(94)	<1(94)	<1(100)
Paracalanidae (0)	3926	8226	5224	8075	3883	6665	3985	5713	2905	5101
% (FO)	38.2(100)	37.3(100)	46.4(100)	39.1(100)	42.0(100)	32.9(100)	52.6(100)	33.0(100)	47.6(100)	31.7(100)
Pontellidae (–)	269	98	243	173	153	281	118	172	123	142
% (FO)	2.6(100)	<1(88)	2.2(100)	<1(100)	1.7(100)	1.4(100)	1.6(100)	1.0(100)	2.0(100)	<1(100)
Pseudodiaptomidae (0)	182	173	334	249	186	199	138	60	43	56
% (FO)	1.8(88)	<1(88)	3.0(92)	1.2(100)	2.0(93)	1.0(100)	1.8(93)	<1(75)	<1(88)	<1(57)
Temoridae (0)	6	27	29	5	28	58	21	72	36	57
% (FO)	<1(38)	<1(21)	<1(62)	<1(44)	<1(67)	<1(69)	<1(40)	<1(69)	<1(65)	<1(71)
Tortanidae (–)	206	46	107	69	100	167	53	75	47	32
% (FO)	2.0(94)	<1(75)	1.0(100)	<1(100)	1.1(93)	<1(100)	<1(87)	<1(100)	<1(76)	<1(93)
Oithonidae (+)	420	5319	541	6092	384	3993	444	3328	307	4359
% (FO)	4.1(100)	24.1(100)	4.8(100)	29.5(100)	4.2(100)	19.7(100)	5.9(100)	19.2(100)	5.0(100)	27.1(100)
Corycaidae (–)	169	43	141	62	140	235	94	197	185	149
% (FO)	1.6(100)	<1(54)	1.2(92)	<1(94)	1.5(100)	1.2(94)	1.2(100)	1.1(100)	3.0(100)	<1(93)
Macrochironidae (0)	12	38	27	53	9	54	35	22	6	45
% (FO)	<1(31)	<1(42)	<1(54)	<1(63)	<1(33)	<1(81)	<1(33)	<1(44)	<1(29)	<1(43)
Clausidiidae (+)	8	54	4	31	-	11	3	11	-	13
% (FO)	<1(19)	<1(63)	<1(15)	<1(25)	-	<1(25)	<1(7)	<1(19)	-	<1(14)
Clytemnestrinae (0)	23	24	26	26	21	35	20	28	12	74
% (FO)	<1(50)	<1(38)	<1(69)	<1(56)	<1(47)	<1(56)	<1(33)	<1(50)	<1(53)	<1(43)
Euterpinidae (0)	751	398	1001	576	722	617	303	595	200	452
% (FO)	7.3(100)	1.8(96)	8.9(100)	2.8(100)	7.8(93)	3.0(100)	4.0(93)	3.4(94)	3.3(100)	2.8(100)
Ectinosomatidae (+)	2	15	-	4	2	6	-	70	-	88
% (FO)	<1(6)	<1(21)	-	<1(19)	<1(7)	<1(38)	-	<1(63)	-	<1(57)
Cirripedia (0)	440	2717	255	278	470	376	278	328	102	222
% (FO)	4.3(94)	12.3(100)	2.3(92)	1.3(100)	5.1(67)	1.9(100)	3.7(93)	1.9(94)	1.7(76)	1.4(100)
Luciferidae (–)	83	4	42	17	98	43	90	39	83	65
% (FO)	<1(100)	<1(83)	<1(100)	<1(100)	1.1(100)	<1(88)	1.2(100)	<1(94)	1.4(100)	<1(100)
Sergestidae (0)	41	51	33	83	42	30	53	113	42	36
% (FO)	<1(100)	<1(75)	<1(100)	<1(88)	<1(100)	<1(63)	<1(100)	<1(88)	<1(100)	<1(93)
Euphausiidae (0)	2	8	1	21	19	17	8	50	26	72
% (FO)	<1(38)	<1(29)	<1(46)	<1(50)	<1(67)	<1(50)	<1(73)	<1(88)	<1(88)	<1(86)
Mysida (–)	4	4	15	3	11	1	13	1	3	3
% (FO)	<1(63)	<1(17)	<1(92)	<1(19)	<1(53)	<1(13)	<1(87)	<1(19)	<1(76)	<1(14)
Pinnotheridae (0)	58	62	18	20	25	19	3	28	5	31
% (FO)	<1(81)	<1(75)	<1(77)	<1(75)	<1(73)	<1(81)	<1(80)	<1(88)	<1(71)	<1(86)
Mityridae (0)	27	1	2	2	23	2	105	7	6	1
% (FO)	<1(31)	<1(8)	<1(8)	<1(25)	<1(60)	<1(38)	1.4(20)	<1(50)	<1(24)	<1(21)
Sesarmidae (–)	345	9	77	5	133	6	157	7	256	13
% (FO)	3.4(75)	<1(50)	<1(85)	<1(44)	1.4(60)	<1(63)	2.1(60)	<1(88)	4.2(76)	<1(71)
Diogenidae (0)	3	9	1	15	8	75	4	31	4	50
% (FO)	<1(25)	<1(54)	<1(31)	<1(88)	<1(67)	<1(81)	<1(27)	<1(75)	<1(29)	<1(93)
Portunidae (0)	1	7	1	9	4	13	1	6	<1	2
% (FO)	<1(81)	<1(79)	<1(77)	<1(100)	<1(73)	<1(88)	<1(60)	<1(94)	<1(53)	<1(71)
Upogebiidae (–)	5	<1	2	-	8	<1	10	<1	11	-
% (FO)	<1(69)	<1(4)	<1(62)	-	<1(80)	<1(6)	<1(87)	<1(13)	<1(88)	-
Ostracoda (0)	<1	-	3	3	13	5	22	37	91	40
% (FO)	<1(31)	-	<1(46)	<1(25)	<1(53)	<1(31)	<1(60)	<1(44)	<1(82)	<1(57)
Chaetognatha (0)	365	274	238	186	191	302	231	200	262	183
% (FO)	3.6(100)	1.2(100)	2.1(100)	<1(100)	2.1(100)	1.5(100)	3.0(100)	1.2(100)	4.3(100)	1.1(100)
Nereididae (0)	2	14	2	51	6	54	1	29	3	41
% (FO)	<1(13)	<1(21)	<1(15)	<1(38)	<1(60)	<1(38)	<1(7)	<1(44)	<1(18)	<1(64)
Sabellariidae (0)	109	149	113	103	49	38	73	26	104	35
% (FO)	1.1(81)	<1(54)	1.0(69)	<1(88)	<1(73)	<1(56)	1.0(80)	<1(81)	1.7(71)	<1(64)
Spionidae (0)	34	164	22	62	43	16	26	83	24	89
% (FO)	<1(44)	<1(58)	<1(31)	<1(75)	<1(67)	<1(63)	<1(47)	<1(63)	<1(47)	<1(71)
Terebellidae (0)	444	100	367	94	18	94	31	59	13	30
% (FO)	4.3(31)	<1(54)	3.3(31)	<1(56)	<1(33)	<1(56)	<1(40)	<1(81)	<1(35)	<1(57)
Polychaete sp. (+)	-	117	-	150	-	60	-	158	-	147

(continued on next page)

Table 3 (continued)

Station	I		C1		C2		C3		C4	
Sampling period	Before	After	Before	After	Before	After	Before	After	Before	After
n	16	24	13	16	15	16	15	16	17	14
% (FO)	-	<1(33)	-	<1(13)	-	<1(13)	-	<1(31)	-	<1(43)
Gastropoda (0)	156	248	197	177	466	195	121	265	117	214
% (FO)	1.5(100)	1.1(83)	1.8(92)	<1(94)	5.0(100)	1.0(94)	1.6(100)	1.5(100)	1.9(100)	1.3(100)
Bivalvia (0)	274	245	333	203	275	262	190	498	143	456
% (FO)	2.7(81)	1.1(79)	3.0(92)	1.0 (88)	3.0(73)	1.3 (81)	2.5(93)	2.9(100)	2.3(94)	2.8(93)
Holothuroidea (0)	111	14	7	35	68	144	4	128	20	267
% (FO)	1.1(25)	<1(21)	<1(23)	<1(19)	<1(20)	<1(31)	<1(27)	<1(50)	<1(18)	1.7(43)
Ophiuroidea (0)	19	230	59	252	45	266	9	335	12	157
% (FO)	<1(13)	1.0(46)	<1(38)	1.2(63)	<1(40)	1.3(63)	<1(33)	1.9(75)	<1(35)	1.0(79)
Hydromedusa (0)	47	179	51	109	43	1369	31	958	9	389
% (FO)	<1(100)	<1(92)	<1(92)	<1(88)	<1(73)	6.8(88)	<1(80)	5.5(100)	<1(88)	2.4(100)
Siphonophora (0)	7	4	9	13	11	39	10	16	13	15
% (FO)	<1(63)	<1(46)	<1(54)	<1(75)	<1(73)	<1(88)	<1(67)	<1(81)	<1(59)	<1(93)
Salpida (+)	-	-	-	-	-	2	<1	5	16	640
% (FO)	-	-	-	-	-	<1 (25)	<1 (7)	<1 (44)	<1 (6)	4.0 (36)
Oikopleuridae (+)	103	1748	80	1987	170	2179	92	1905	88	1183
% (FO)	1.0(63)	7.9(96)	<1(62)	9.6(94)	1.8(80)	10.8(88)	1.2(73)	11.0(94)	1.4(65)	7.4(100)
Bryozoa (0)	23	46	25	48	10	4	11	41	18	8
% (FO)	<1(50)	<1(50)	<1(69)	<1(44)	<1(40)	<1(13)	<1(27)	<1(44)	<1(24)	<1(14)
Phoronidae (0)	10	81	16	29	5	3	4	5	1	34
% (FO)	<1(25)	<1(25)	<1(46)	<1(13)	<1(13)	<1(19)	<1(13)	<1(13)	<1(6)	<1(36)
S	71	73	72	71	69	78	75	80	76	74
Total zooplankton abundance	10,266	22,031	11,264	20,644	9235	20,254	7580	17,336	6104	16,074

pairs (I and Cs) and between before–after pairs due to the Kapar power plant ($p < 0.01$, Table 5). There were also significant interactions in the measure of community structure between BA and I vs. Cs, suggesting differences in community structure due to stations as well as before and after impact (PERMANOVA, $p < 0.01$; Table 5). The community structure of zooplankton at Stn I was significantly different from Stn Cs before impact (PERMANOVA, $p < 0.05$) as well as after impact (PERMANOVA, $p < 0.01$; Table 5). However, the similarity of zooplankton community between Stn I and Stn Cs was lower after impact (59.9%) than before impact (65.6%) (Table 5). The before–impact community at Stn I and Stn Cs encountered a significant change in structure after impact ($p < 0.01$), with a lower similarity index computed for Stn I (59.1%) as compared to Stn Cs (61.9%) after impact (Table 5).

As indicated by significant interactions between BA and Cs (PERMANOVA, $p < 0.05$), the zooplankton community structure was also significantly different among controls and between before and

after impact (Table 5). The community at Stn C1 was significantly different from the other three controls before impact, with similarity between them that decreased from Stn C2 (64.6%) to Stn C4 (63.7%) (Table 5). Before impact, the community structure from Stn C2 to Stn C4 was statistically similar ($p > 0.05$, Table 5). The community structure at Stn C1 and Stn C2 was similar after impact (PERMANOVA, $p > 0.05$) and their community structure was significantly different from that of Stn C4 (PERMANOVA, $p < 0.05$; Table 5). All controls showed a significant change in community structure from before–impact to after–impact conditions (PERMANOVA, $p < 0.01$; Table 5). The lower similarity computed for Stn C4 (61.3%) suggested a more dramatic change encountered at this station compared to the other three controls (Table 5).

The ordination biplot in Fig. 4 from PCO visualizes the differences in zooplankton community structure between impact and control stations (spatial), and between before and after impact effects (time). The first

Table 4
Results of ANOVA tests on diversity measures and abundance of key zooplankton taxa.

Variable	Combined tests			Repartitioned tests					
				Between impact and controls (I vs. Cs)			Among controls (Cs)		
	Before–after (BA)	Stations (S)	Interactions (BA x S)	Before–after (BA)	Impact–Controls (I vs. Cs)	Interactions (BA x I vs. Cs)	Before–after (BA)	Controls (Cs)	Interactions (BA x Cs)
df	1	4	4	1	1	1	1	3	3
<i>Diversity measures</i>									
J'	ns	ns	ns	ns	ns	ns	ns	ns	ns
H'	ns	ns	ns	ns	ns	ns	ns	ns	ns
<i>Abundance</i>									
Total zooplankton	B < A **	*	ns	B < A**	ns	ns	B < A**	ns	ns
Paracalanidae	B < A*	ns	ns	B < A*	ns	ns	B < A*	ns	ns
Oithonidae	B < A**	ns	ns	B < A**	ns	ns	B < A**	ns	ns
Acartiidae	B > A**	**	ns	B > A**	I > Cs**	ns	B > A*	C1 ^{a,b} C2 ^a C3 ^{a,b} C4 ^b **	ns
Centropagidae	B < A**	**	ns	B < A*	I < Cs**	ns	B < A**	ns	ns
Eucalanidae	ns	**	**	ns	I < Cs**	**	B < A**	ns	ns
Euterpinae	ns	ns	ns	ns	ns	*	ns	ns	ns
Cirripedia	B < A**	**	ns	B < A**	I > Cs**	ns	B < A**	ns	ns
Luciferidae	B > A**	*	ns	B > A**	ns	**	B > A*	ns	ns
Chaetognatha	ns	ns	ns	ns	ns	ns	ns	ns	ns
Oikopleuridae	B < A**	ns	ns	B < A**	ns	ns	B < A**	ns	ns

Abbreviations: J' Pielou's evenness, H' Shannon–Wiener diversity index, B before impact, A after impact, I impacted station, C1–C4 control stations, Cs control stations combined, ns not significant, * $p < 0.05$, ** $p < 0.01$.

two axes (PCO1 and PCO2) accounted for 32.1% of the total variability in zooplankton (45 taxa) abundance. With few exceptions, the horizontal axis or PCO1 defines the before-impact stations (empty symbols) and zooplankton community (arrows) along the right or positive axis, and the after-impact stations (filled symbols) and zooplankton community (arrows) along the left half or negative axis of PCO1. The vertical axis or PCO2 generally defines the spatial gradient of zooplankton community from farther off shore (negative axis) to near shore (positive axis). Samples from the impacted Stn I (filled circles, I) were mainly positioned on the top-left quadrant of the biplot, while samples from Stn C4 (filled squares, 4) were mainly positioned on the bottom-left quadrant. Samples from Stn C1 (filled squares, 1), Stn C2 (filled squares, 2) and Stn C3 (filled squares, 3) lay between Stn I and Stn C4. During before-impact conditions, the samples of Stn I (empty circles, I) were close to those of Stn C1 (empty squares, 1) along the positive axis of PCO2 (top-right quadrant), while Stn C4 (empty squares, 4) were positioned mainly along the negative axis of PCO2 (bottom-right quadrant). The PCO ordination biplot conformed to the results generated by the PERMANOVA tests.

Based on the taxa-specific abundance data (see Table 3), the 45 selected zooplankton taxa subjected to PCO analysis can be divided into three groups based on significance test between before and after impact abundance (t-test, $p < 0.05$): (1) “Vulnerable” – those that had encountered a significant reduction in abundance at Stn I after impact, (2) “Resilient” – those that had not encountered any significant change in abundance at Stn I after impact, and (3) “Opportunistic” – those that displayed an exceptional or significant increase in abundance after impact.

The PCO analysis revealed distinct spatial and time differences among the three groups. In particular, the vulnerable group of taxa (with yellow highlights) except “Euca” is set apart from the resilient group (without highlights) and opportunistic group (with purple highlights). All vulnerable taxa which together totaled 11 taxa were crustaceans comprising three subgroups: (i) mysids (Mysd) and decapods Upogebiidae (Upog), Luciferidae (Luci) and Sesarminae (Sesar) which were considered ubiquitous 30 years ago but now present in low numbers. For instance, Upogebiidae was frequently sampled before impact (>60% of occurrence) but was almost absent after impact (<1%

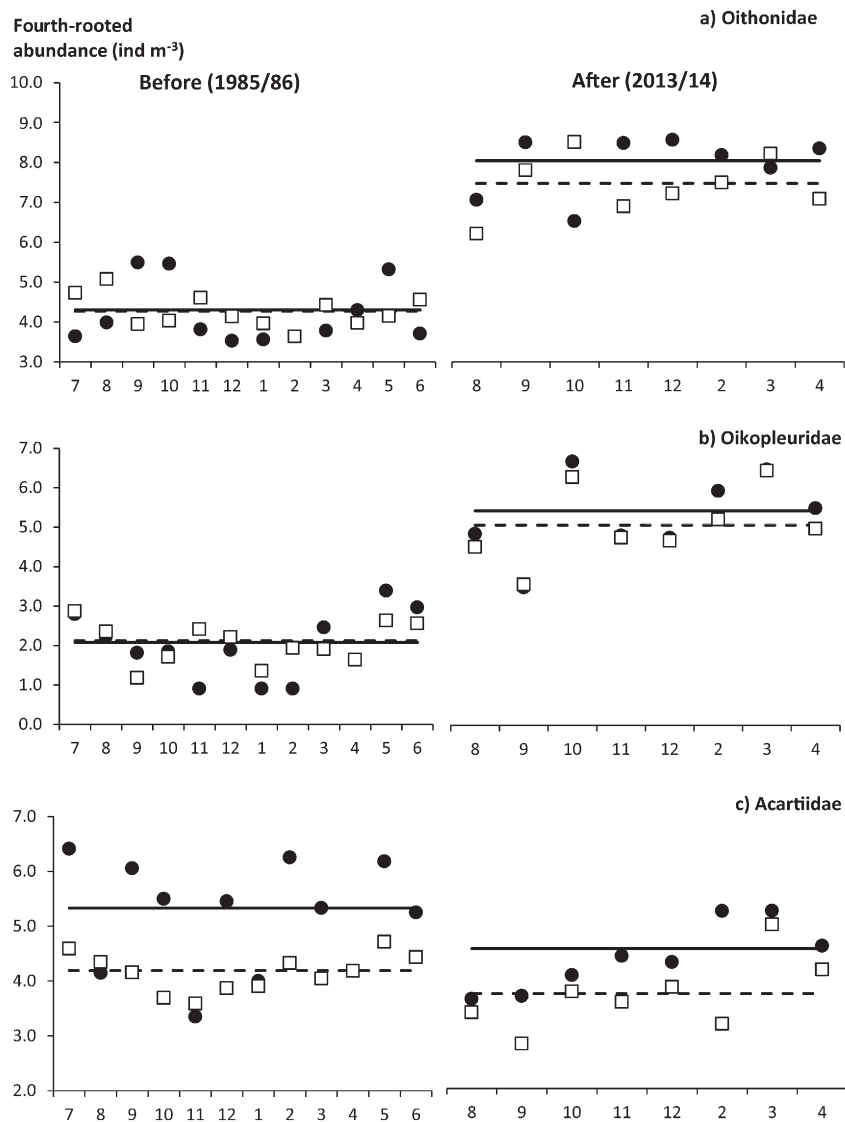


Fig. 3. Monthly anomalies of key zooplankton taxa at impacted (filled circles) and control (empty squares) sites in Klang Strait before (left panel) and after (right panel) establishment of Kapar power station. Vertical axis indicates anomaly (or abundance) in term of the number of standard deviation from the mean of all sampling months drawn as solid horizontal line (impacted site, I) or dotted horizontal line (control sites, Cs) on its fourth-root transformed value. Horizontal axis indicates the month of sampling (e.g. 7 = July).

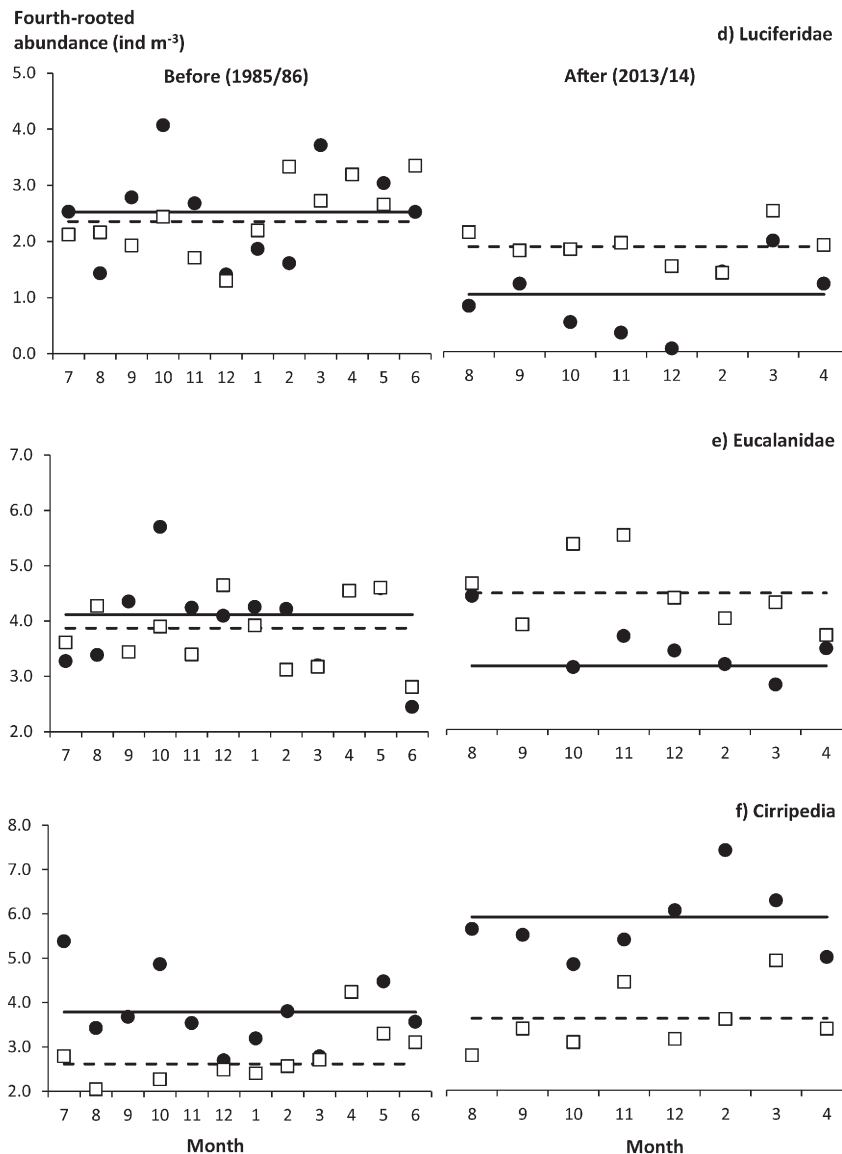


Fig. 3 (continued).

occurrence, see Table 3); (ii) calanoid copepods Acartiidae (Acar) and Tortanidae (Tor) which were more confined to coastal waters (i.e. I, C1 and C2) had decreased in abundance over the past 30 years; (iii) calanoid copepods Eucalanidae (Euca), Pontellidae (Pont), Calanidae (Cal), Euchaetidae (Euch) and the poecilostomatoid copepod Corycaidae (Cory) which were quite homogenous in abundance at all stations before impact were very much reduced at Stn I after impact. However, their abundances at the control stations remained quite similar or even increased after impact (see Table 3).

There were altogether 28 taxa considered as belonging to the resilient group which comprised the following four subgroups: (i) seven taxa belonging to the calanoid copepod Pseudodiaptomidae (Pseu), the poecilostomatoid Macrochironidae (Macro), the harpacticoid Clytemnestrinae (Cly), the Mityridae decapod larvae (Mity), the Chaetognatha (Chae) and Bryozoa (Bry) and Phoronidae (Phoro) larvae, which showed no significant difference in abundance before and after impact at both impacted and control stations, but all were rather confined to shallow coastal waters (Stn I, C1 and C2); (ii) four taxa that included Euphausiidae (Euph), Ostracoda (Ost), the Holothuroidea echinoderm larvae (Holo) and Siphonophora (Sipho), which were not

significantly different before and after impact at both impacted and control stations, but were more confined to deeper waters (C3 and C4); (iii) the calanoid copepod Temoridae (Temo), the harpacticoid copepod Euterpinae (Eut), the decapod larvae from the families of Sergestidae (Ser), Pinnotheridae (Pin) and Portunidae (Port) and the polychaete larvae from the families of Sabellariidae (Sabe) and Terebellidae (Tere). All were homogeneously found at all stations and there was no change in their abundance before and after impact; (iv) 13 taxa including the calanoid copepods Paracalanidae (Para) and Centropagidae (Cen), naupliar Cirripedia (Cirri), anomuran zoea of Diogenidae (Dio), Gastropoda (Gas) and Bivalvia (Bv), polychaete larvae from the families Nereidae (Nerei) and Spionidae (Spio), the Ophiuroidea echinoderm larvae (Ophiu) and the hydromedusa (Hydro), which increased in abundance after impact. The small copepod Paracalanidae was significantly higher in abundance at all stations, while the cirriped nauplius thrived at the impacted Stn I. The remaining taxa did not show significant change in abundance before and after impact at Stn I but were more abundant at control stations after impact.

The opportunistic group comprised six taxa including the cyclopoid copepod Oithonidae (Oitho), the poecilostomatoid Clausidiidae (Clau),

Table 5

Results of PERMANOVA tests on zooplankton community structure in Klang Strait before and after Kapar power plant establishment.

Variable	Combined tests			Repartitioned tests						
				Between impact and controls			Among controls			
	Before–After (BA)	Stations (S)	Interactions (BA x S)	Before–after (BA)	Impact–controls (I vs. Cs)	Interactions (BA x I vs. Cs)	Before–after (BA)	Controls (Cs)	Interactions (BA x Cs)	
df	1	4	4	1	1	1	1	3	3	
Pseudo-F	16.97	3.25	1.75	13.98	6.60	2.73	13.47	2.15	1.42	
p (perm)	0.00**	0.00**	0.00**	0.00**	0.00**	0.00**	0.00**	0.00**	0.03*	
Average similarity matrix										
BA x I vs. Cs										
Level	I			Cs						
	Before	Before	After	Before	Before	After				
	After	59.1**	61.1	After	61.9**	65.7				
Level	Before			After						
	I	I	Cs	I	I	Cs				
	Cs	65.6*	65.0	Cs	59.9**	65.7				
BA x Cs										
Level		C1	C2	C3	C4					
	Before	67.6	64.3	65.4	65.9					
	After	65.6	66.6	67.2	66.1					
	BA	62.9**	62.8**	62.0**	61.3**					
Level	Before					After				
		C1	C2	C3	C4		C1	C2	C3	C4
	C1	67.6					65.6			
	C2	64.6**	64.3				65.6	66.6		
	C3	65.2**	64.8	65.4			65.5*	66.2	67.2	
	C4	63.7**	64.7	65.8	65.9		64.1**	64.8*	66.8	66.1

Boldface denotes significance at * p < 0.05 and ** p < 0.01.

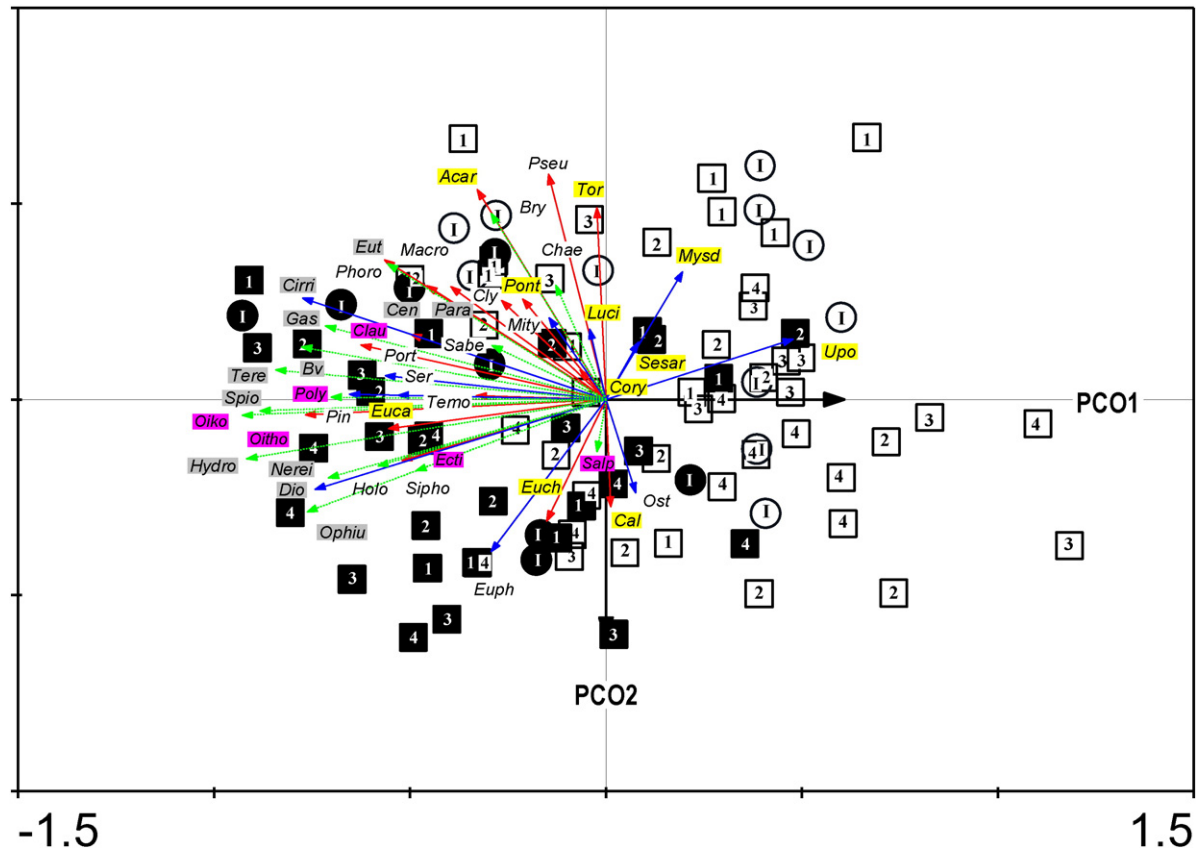


Fig. 4. PCO ordination biplot of 45 zooplankton taxa and monthly-sampled stations before (July 1985–June 1986) and after power plant impact (August 2013–April 2014) in Klang Strait. Alphabetical letter: I denotes impacted station; numerical letters from 1 to 4 denote control stations from C1 to C4. Symbols: ○ Stn I before impact, ● Stn I after impact, □ control stations before impact, ■ control stations after impact. Copepods (red arrows): *Acar* Acartiidae, *Cal* Calanidae, *Cent* Centropagidae, *Cory* Corycaidae, *Clau* Clausidiidae, *Ecti* Ectinosomatidae, *Euca* Eucalanidae, *Euch* Euchaetidae, *Eut* Euterpinidae, *Macro* Macrochironidae, *Oitho* Oithonidae, *Para* Paracalanidae, *Pseu* Pseudodiaptomidae, *Pont* Pontellidae, *Temo* Temoridae, *Tort* Tortanidae; non-copepod crustaceans (blue arrows): *Cirri* Cirripedia, *Dio* Diogenidae, *Luci*, Luciferidae, *Mity* Mityridae, *Mysd* Mysida, *Ost* Ostracoda, *Sesar* Sesarminae, *Upo* Upogebiidae; non-crustacean zooplankton (green dotted arrows): *Bry* Bryozoa, *Chae* Chaetognatha, *Euph* Euphausiidae, *Holo* Holothuroidea, *Hydro* Hydromedusae, *Nerei* Nereididae, *Oiko* Oikopleuridae, *Ophiu* Ophiuroidea, *Poly* unidentified polychaete larvae, *Phoro* Phoronidae, *Sabe* Sabellariidae, *Salp* Salpida, *Sipho* Siphonophora, *Spio* Spionidae, *Tere* Terebellidae.

the harpacticoid Ectinosomatidae (Ecti), the unidentified polychaete larvae (Poly), the appendicularian Oikopleuridae (Oiko) and the tunicate Salpida (Salp). The Oithonidae and Oikopleuridae were ubiquitous organisms along the sampling transect. These taxa exhibited at least a tenfold increase in abundance at all stations from before to after impact conditions. The copepods Clausidiidae and Ectinosomatidae, the unidentified polychaete larvae and the salps were comparatively rare or absent before impact but appeared to be common and abundant after impact (see Table 3).

4. Discussion

4.1. Zooplankton abundance and community structure: resilient, vulnerable and opportunistic groups

The present study reports for the first time the dramatic changes that occurred in a tropical zooplankton community in Klang Strait after nearly 30 years of operation of KPS. The most striking finding of the present study was the replacement of the large crustacean zooplankton by small crustacean and non-crustacean zooplankton (see Fig. 4). Such changes in the zooplankton community and other events (see below) are symptomatic of warming effects (Richardson, 2008; Sheridan and Bickford, 2011; Rice et al., 2015). The stark changes in Klang Strait parallel the striking manifestation of zooplankton response to global warming in northeast Atlantic waters, that is, the replacement of cold-water copepod assemblages (dominated by the large-bodied *Calanus finmarchicus*) by warm-water copepod assemblages (dominated by small-bodied *Calanus helgolandicus*) (Beaugrand et al., 2002). Similarly, estuarine copepods dominated by *Acartia* spp. became smaller in size, the proportion of the small copepod *Oithona* sp. increased and the two largest-sized copepod genera (*Metridia* and *Candacia*) had disappeared in Long Island Sound (USA) after more than 50 years of warming calculated at the rate of $0.03\text{ }^{\circ}\text{C y}^{-1}$ (Rice et al., 2015). Structural change in the zooplankton community is the most intense at the immediate vicinities of KPS spreading up to 4 km offshore of it, indicating that the power station causes or exacerbates the environmental impacts impinging on the zooplankton community. The environmental impact on Klang Strait's zooplankton community is however not reflected by any change of the standard community ecology metrics measured, including overall species richness and diversity. This is interesting because a similar observation of no change in standard community ecology metrics (species number, alpha diversity and dominance indices) but a clear change in community structure has been documented for fish and crustaceans at a power station in UK (Henderson et al., 2011). In contrast, copepod richness and diversity in the northeast Pacific has been documented to increase considerably since the 1970s, but this is primarily due to increased sea temperature and consequently, the intrusion of warm-water species from the south (Hooff and Peterson, 2006). Nevertheless, intrusions by alien species such as this may not occur in tropical waters where sea temperature is already high and temperature differentials across spatial scales are small (this study). The Klang Strait case thus demonstrates intrinsic changes (due to the impact) within the zooplankton community, a phenomenon that likely occurs in enclosed waters. Hydrodynamic and dispersal modeling of the confined Klang Strait demonstrated that the dominant macro-tides slosh seawater back (ebb) and forth (flood) along its narrow 70-km long channel, and that the strait basically functions as a large retention area for planktonic larvae of penaeid shrimps (Chong et al., 1996). The long term retention of its plankton with limited lateral and longitudinal exchanges outside it (i.e. the Malacca Straits) may have supported such intrinsic changes involving resilient, vulnerable and opportunistic groups in tandem with their vulnerability to the deleterious effects of environmental stressors. It seems that the present contribution of the resilient and opportunistic groups to species richness and diversity is similar to the contribution of the vulnerable groups 30 years ago. This indicates that the deleterious environmental stressors of the

vulnerable taxa could be tolerated by the resilient group or even have a positive effect on the opportunistic group.

The crustaceans Acartiidae and Luciferidae in Klang Strait were among the zooplankton that encountered a stark decrease in abundance since the past 30 years particularly in the area close to KPS (i.e. Stn I and C1). Other vulnerable taxa included Mysida, Upogebiidae, Sesarminae and Tortanidae. The large copepods Calanidae, Eucalanidae, Pontellidae and Corycaidae were found some distance away from KPS after impact and were not affected since the establishment of KPS. Increased SST could explain the vulnerability of the large crustaceans in Klang Strait, since this trend closely parallels the findings from temperate waters. A decline in the dominance of Acartiidae has lately been observed in Tokyo Bay (Uye, 1994), Adriatic Sea (Conversi et al., 2009) and central Chesapeake Bay, USA (Kimmel et al., 2012). In San Francisco Estuary, the abundance of mysids and two dominant copepods *Eurytemora* and *Acartia* dropped significantly from 1972 to 2008 (Winder and Jassby, 2011). The reasons for such declines of zooplankton abundance are not immediately clear since the possible causes include sea temperature rise (Conversi et al., 2009; Kimmel et al., 2012), eutrophication (Uye, 1994; Kimmel et al., 2012), change in food composition and quality (Winder and Jassby, 2011), and predation (Kimmel et al., 2012).

With increasing anthropogenic disturbances, the consistent trend observed across the global marine ecosystems is that Oithonidae outcompetes calanoid copepods (e.g. Uye, 1994; Svensen et al., 2011; Winder and Jassby, 2011). The present study in tropical waters further supports this view. The dominance of Oithonidae has been attributed to (1) successful physiological and behavioral adaptations that include low metabolic and high reproductive rates with increased sea surface temperature (Castellani et al., 2005; Ward and Hirst, 2007; Almeda et al., 2010); (2) their ability to exploit food in the microbial food web (Uye, 1994; Bouley and Kimmerer, 2006; Zamora-Terol et al., 2014), and (3) low predation rate by visual predators due to their less conspicuous small body and low nutritional value (Bouley and Kimmerer, 2006; Svensen et al., 2011). Much less information is available on changes that involved appendicularians, cirripeds and other non-crustacean zooplankton. The available evidence however suggests temperature and organic pollution as causal factors. Highfield et al. (2010) reported that the interannual variability of larval cirripeds, polychaetes, echinoderms and molluscs at Station L4 (Plymouth), UK, were consistent over 20 years and their seasonality was closely linked to temperature and phytoplankton abundance. In an organically-polluted estuary of the Basque coast (Spain), cirriped nauplii, appendicularians and polychaete (Spionidae) larvae which predominated after were not affected by low water quality (Intxausti et al., 2012). Appendicularians were found to co-occur with Oithonidae in high numbers in the eutrophicated Tokyo Bay (Ishii and Tanaka, 2001; Sato et al., 2008). Laboratory experiments have provided further evidence that the growth development of appendicularians and cirriped nauplii is primarily regulated by temperature and food availability (Anil et al., 2001; Troedsson et al., 2013). These results indicate that appendicularians and cirripeds are resilient to perturbations due to temperature rise and eutrophication, and explain why these taxa now thrive in the perturbed Klang Strait.

Another finding from the present study that should be highlighted is the episodic occurrence of salp swarms now observed in Klang Strait. This is another indicator of warming effects on zooplankton (Richardson, 2008). Salps were almost absent 30 years ago but they heavily clogged the plankton net at Stn C4 in April 2014 (see Table 3). Although several studies reported the interannual and demographic variability of salps (e.g. Licandro et al., 2006; Henschke et al., 2014), the exact factors that initiated salp swarms have not been confirmed (Hay, 2006; Licandro et al., 2006). Time series data however indicate that the interannual salp outbreaks always coincided with positive temperature anomaly, and that the outbreak frequency has increased in recent years (Hay, 2006; Licandro et al., 2006). Thus, salp outbreaks in Klang Strait are also likely due to elevated SST or the result of hydro-

climatic interactions that coincided with a prolonged drought during the first half of 2014.

4.2. Potential environmental drivers of shift in zooplankton community structure

4.2.1. Sea temperature

Rise in SST was evident from the higher mean SST recorded in recent years as compared to 30 years ago and earlier in the mid-70s (see Fig. 2). Based on all temperature differences measured before (1985/86) and after impact (2013/14), the overall mean SST in Klang Strait has risen by 0.58 °C or at the rate of 0.19 °C per decade. In particular, SST at the controls (excluding Stn I) has risen by 0.57 °C or 0.19 °C per decade. If diurnal temperature variation is considered (i.e. by taking the same time segment from 1 PM to 6 PM), the mean SST recorded by the HOBO pendant at KPS (31.15 ± 0.49 °C) was 1.63 °C higher than the mean SST at KPS before impact (29.52 ± 1.10 °C). This is a rise of 0.54 °C per decade. In comparison, the SST in the northern Atlantic has risen 1.13 °C since 1962 or 0.28 °C per decade due to climate change (Wiltshire and Manly, 2004). Based on the IPCC (2014)'s projection, the global mean SST has increased by 0.11 °C per decade. Thus, the SST rise per decade at controls in Klang Strait is between the northern Atlantic and global mean SST rise, while the SST rise at KPS immediate waters is about five times higher than the global rate of SST rise. Assuming that the thermal effluent effects due to KPS completely dissipated at controls (see Anton, 1990), then the SST rise at controls is an indication of climate change. Given the small incremental change in SST in Klang Strait over time, it may be difficult to associate the small increase in temperature to changes in the zooplankton community. Laboratory experiments however showed significant negative correlation between critical thermal maximum and body length of copepods suggesting that temperature increase will reduce diversity in a copepod community that will also tend to comprise of small-sized species (Jiang et al., 2009). Nonetheless, field data over three decades since KPS was established have shown that the change in zooplankton community is not only dramatic but mirrored similar changes in temperate waters (described above) that are ascribed to a small rise in mean SST, albeit over a longer period of more than five decades (Richardson, 2008; Mackas et al., 2012; Rice et al., 2015).

4.2.2. PAH and pH

The measured PAH contamination from KPS throughout our sampling period was far below the chronic level of $<7000 \mu\text{g L}^{-1}$ (Barata et al., 2005; Saiz et al., 2009), and is therefore not expected to exert significant impact on the zooplankton community. The overall pH in Klang Strait has dropped by 0.37 from the past 30 years, which is a much larger rate of decline compared to oceanic waters (0.2 to 0.4) as predicted by IPCC in 100 years (IPCC, 2014). Apart from atmospheric CO₂ increase, decrease in pH can also be induced by anthropogenically-released chemical compounds. Coal burning and use of sodium hypochlorite biocide by the power station as well as riverine discharges from the mainland may have reduced the pH of Klang Strait. The lowest reduced pH at 7.47 (Stn I) is not expected to duly affect the mostly copepod zooplankton in the short term (Yamada and Ikeda, 1999; McConville et al., 2013), except perhaps the Acartidiidae. The reproductive success and naupliar survival of *Acartia tonsa* are reported to be negatively affected by reduced pH from ca. 8.2 ($p\text{CO}_2 = 385 \mu\text{atm}$) to 7.8 ($p\text{CO}_2 = 1000 \mu\text{atm}$) (Cripps et al., 2014a,b). Nonetheless, the chronic effects of lower pH on marine zooplankton are not so well studied.

4.2.3. Eutrophication

Monthly variations in phytoplankton biomass (chl. *a*) in Klang Strait were very large in the present study; *Skeletonema* blooms often broke out, followed by large blooms of pinkish bacteria that presumably utilize them. Similarly, episodes of peak gross primary production followed by peak heterotrophic bacterial production in Klang Strait had been

reported by Lee and Bong (2006). Blooms of *Microcystis* also regularly clogged the fine-mesh plankton nets on sampling occasions. However, *Microcystis* blooms were not observed 30 years ago. Although hypoxia due to eutrophication was not detected during this study (DO generally $>4 \text{ mg L}^{-1}$), episodes of hypoxia ($\text{DO} < 125 \mu\text{M}$ or $< 4 \text{ mg L}^{-1}$) occurring in the narrow channels of southern Klang Strait had been reported by Lee and Bong (2006). Thus, the microbial-based food web may have gained some measure of importance in Klang Strait after 30 years of anthropogenic perturbation. The present nutrient levels of phosphate and nitrate measured in the main channel of Klang Strait are comparable to those reported by Lee and Bong (2006) who had earlier concluded that southern Klang Strait waters are eutrophic. Their reported nutrient concentrations included NH_4^+ $11.76 \pm 9.06 \mu\text{M}$, NO_3^- $2.78 \pm 1.99 \mu\text{M}$, NO_2^- $2.39 \pm 1.93 \mu\text{M}$, and PO_4^{3-} up to $5.68 \mu\text{M}$. Lee and Bong (2012) further reported that Klang Strait had experienced a dramatic increase in total suspended solids (TSS) from 1990 to 2003 that currently exceeds 130 mg L^{-1} , attributable to land clearing activities. They suggested that although net primary production was relatively high ($9.7 \pm 3.1 \mu\text{M C h}^{-1}$), it was likely depressed by high TSS, and that primary production alone could not have supported the high bacterial carbon demand ($11.3 \pm 6.8 \mu\text{M C h}^{-1}$) without allochthonous inputs. Viral lysis and protistan bacterivory however accounted for 20% of the bacterial production in Klang Strait, suggesting that part of it may be consumed by benthic filter feeders (Lee and Bong, 2012) and zooplankton. Bacterial production and the shift in contribution of phytoplankton species in the food web of Klang Strait appear to favor zooplankters with low metabolic rates as similarly reported in other waters (Castellani et al., 2005; Ward and Hirst, 2007; Almeda et al., 2010). This may explain the dramatic shifts in abundance of Oithonidae and Oikopleuridae in Klang Strait.

4.2.4. Others

Hypochlorite or residual chlorine around KPS waters was not measured in the present study. However, the effect of chlorine and its byproducts (organohalogenes) on biota in the receiving waters is expected to be localized (Rajagopal et al., 2003; Bamber and Seaby, 2004; Poornima et al., 2005; Taylor, 2006). Interestingly, the use of hypochlorite biocide to reduce barnacle fouling in power plants appears to have no effect on cirriped larvae found abundantly at KPS vicinities. Since increased temperature shortens the development of cirriped larvae (Qiu and Qian, 1997; Anil et al., 2001), low pH level of 7.4 does not duly affect larval cirriped development and attachment (McDonald et al., 2009), and given the abundant *Skeletonema* food favored by cirriped larvae (Qiu and Qian, 1997), the current water quality in Klang Strait actually favors the observed proliferation of cirriped larvae at KPS. The mean concentrations ($\mu\text{g L}^{-1}$) of eight heavy metals As (24.70), Cu (2.51), Cr (8.77), Cd (0.48), Ni (2.39), Pb (8.00), Hg (0.02) and Zn (52.90) in Klang Strait indicate that heavy metal pollution is not yet considered hazardous (Sany et al., 2013). Contaminated waters are more in the southern Klang Strait due to agro-industrial discharges via Klang River. Entrainment mortality as a function of temperature difference, biocide effect, mechanical injury, passage time, plankton characteristics, etc. can directly kill off all or almost all zooplankton that passes through the power plant's cooling-water system (Carter and Reader, 2009). However, there is no information on the scale of entrainment mortality in KPS. Since zooplankton species respond differently to entrainment and thermal stress effects, the high and variable mortalities of zooplankton could contribute to changes in species abundance and hence, community structure.

4.3. Food web changes and implications

Climate change and anthropogenic stressors can synergistically accelerate changes in ecosystem structure via the food web (e.g. Li et al., 2009; Möllmann et al., 2008; Suikkanen et al., 2013). Climate change that leads to increased SST, precipitation and dissolved inorganic

nutrients can lead to a chain of events that includes eutrophication, replacement of the large more nutritious diatoms by small less nutritious (and even toxic) microalgae, and the proliferation of bacteria, flagellates and ciliates that are favored as food by small copepods such as the Oithonidae (Svensen and Kiørboe, 2000; Bouley and Kimmerer, 2006; Zamora-Terol et al., 2014). Any increase in the density of fine food particles channeled through the microbial loop will favor the proliferation of the filter-feeding appendicularians and salps in eutrophicated environments. As in Klang Strait, the dominance of Oithonidae and gelatinous zooplankton in many anthropogenically-impacted ecosystems (Hay, 2006; Rice et al., 2015) may indicate a shift in importance of the microbial food web, but believed to be less energy efficient than the classical phytoplankton-based food web (Suikkanen et al., 2013). Chew et al. (2012) found that the contribution of Oithonidae to juvenile fish nutrition in tropical mangrove estuaries was much less than large-bodied zooplankton such as the calanoid Pseudodiaptomidae, mysids and *Acetes*. Given the sharp increase in abundance of Oithonidae in Klang Strait since the past 30 years, the changing environmental conditions have currently favored the Oithonidae especially *O. attenuata* and *O. simplex*. This shift in abundance of the type of favored copepod species as food for fish larvae can have repercussion in fisheries as exemplified in temperate waters (Beaugrand et al., 2002).

It is possible that the change in zooplankton community structure results from the change in their phytoplankton food in Klang Strait. Phytoplankton appeared to be previously dominated by large, solitary or centric diatoms such as *Biddulphia*, *Coscinodiscus*, *Cyclotella* and *Ditylum* thirty years ago (Chong, unpublished data). However, since the prevailing view suggests stronger consumer-driven control due to warming (O'Connor et al., 2009), it is likely that the larger, more favored diatoms are being rapidly consumed (Hutchins et al., 2007). Although the present phytoplankton community was not quantified, phytoplankton samplings in 1999 indicated the relative dominance of small, needle-like or chained diatoms such as *Rhizosolenia*, *Chaetoceros*, *Skeletonema*, *Thalassiothrix* and *Nitzschia* in Klang Strait (Aishah, 2012).

An interesting example of what appears to be the cascading effects of warming via trophic interactions among zooplankters and predators is also observed in Klang Strait. The increase in Oikopleuridae will also increase their discarded tests, which are important food for the harpacticoid *Microsetella* and some poecilostomatoid copepods (Koski et al., 2007). Hence, the increased abundance of Ectinosomatidae (*Microsetella norvegica*) and poecilostomatoid copepods from the past 30 years could occur in tandem with the increased discards of Oikopleuridae tests in Klang Strait.

4.4. Limitations of study

This study is limited by the available samples that were archived in our zooplankton collection prior to KPS operations. Although the before-impact samples covered a wide area of Klang Strait, these samples were taken north of KPS, and only few samples were taken south of KPS. For this reason, the BACI control stations cannot extend south of KPS. Hence, the present analysis could not determine for certain that KPS's operations and major impacts (thermal effluents and entrainment) were the only or key factors responsible for the changes that occurred in the zooplankton community. As a result, this study had to take cognizance of other possible effects from nearby harbor activities including the levels of PAH and heavy metals in Klang Strait. If any anthropogenic impacts are of concern and not attributed to KPS, it is likely eutrophication of the nearshore waters. However, the real and long-term effect of eutrophication on the coastal food web is not immediately clear and further studies are desired.

5. Conclusion

Sea warming (due to discharged thermal effluent and climate change) and eutrophication are likely the main anthropogenic stressors

altering the structure of Klang Strait's zooplankton community since the last three decades. Although total community diversity has not changed, the trend effect is a dramatic increase in abundance of small-bodied copepods at the expense of previously dominant large-bodied species, increased abundance of gelatinous salps and larvaceans, and decreased abundance of luciferids, resulting in characteristic taxa of vulnerable, resilient and opportunistic groups. The resilient and opportunistic groups appear to dominate the zooplankton in terms of their sheer numbers. Interestingly, these community changes due to warming of tropical waters parallel those reported for temperate waters, suggesting that tropical zooplankton respond similarly to their temperate counterparts even though the seasonal temperature differential is much smaller in tropical waters. This raises the question of whether most of the vulnerable tropical species of zooplankton are already at their limits of tolerance. It also raises the question of whether the vulnerable species are endemic or true tropical species, while the resilient or opportunistic species are cosmopolitan or even alien imports (e.g. via ship ballasts) that are more tolerant species having acclimated well to tropical conditions. It is possible that warming which increases animal respiration rate also increases food consumption (Almeda et al., 2010) thus disadvantaging the larger zooplankton that preferentially feed on larger more nutritious diatoms, while the smaller zooplankton fortuitously utilize the higher production of bacterioplankton and nanoplankton. In this respect, eutrophication could be a factor that drives the flow of detrital organic matter to the microbial loop of the coastal food web. It is predicted that a continued increase in sea surface temperature whether anthropogenically-induced or due to climate change will see more loss of large-bodied zooplanktons, the preferred food for fish larvae and other meroplankton, with grave consequences to fishery production.

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References

- Aishah, S., 2012. Phytoplankton. In: Sasekumar, A., Chong, V.C. (Eds.), *Mangrove and Coastal Environment of Selangor*, Malaysia Monograph Series 14. Institute of Ocean and Earth Sciences, University of Malaya, pp. 174–203.
- Almeda, R., Augustin, C.B., Alcaraz, M., Calbet, A., Saiz, E., 2010. Feeding rates and gross growth efficiencies of larval developmental stages of *Oithona davisae* (Copepoda, Cyclopoida). *J. Exp. Mar. Biol. Ecol.* 387, 24–35.
- Anderson, M.J., Gorley, R.N., Clarke, K.R., 2008. PERMANOVA + for PRIMER: Guide to Software and Statistical Methods. PRIMER-E, Plymouth, UK.
- Anil, A.C., Desai, D., Khandeparkar, L., 2001. Larval development and metamorphosis in *Balanus amphitrite* Darwin (Cirripedia; Thoracica): significance of food concentration, temperature and nucleic acids. *J. Exp. Mar. Biol. Ecol.* 263, 125–141.
- Anton, A., 1990. Effects of Thermal Discharge on Marine Phytoplankton. In: Phang, S.M., Sasekumar, A., Vikineswary, S. (Eds.), *Proceedings of 12th Seminar Annual Seminar on Research Priorities for Marine Sciences in the 90's*, 18 November, 1989. Institute of Advanced Studies, University of Malaya, Kuala Lumpur, pp. 57–64.
- Azila, A., Chong, V.C., 2010. Multispecies impingement in a tropical power plant, Straits of Malacca. *Mar. Environ. Res.* 70, 13–25.
- Bamber, R.N., 1995. The influence of rising background temperature on the effects of marine thermal effluents. *J. Therm. Biol.* 20, 105–120.
- Bamber, R.N., Seaby, R.M.H., 2004. The effects of power station entrainment passage on three species of marine planktonic crustacean, *Acartia tonsa* (Copepoda), *Crangon crangon* (Decapoda) and *Homarus gammarus* (Decapoda). *Mar. Environ. Res.* 57, 281–294.
- Bamber, R.N., Spencer, J.F., 1984. The benthos of a coastal power station: thermal discharge canal. *J. Mar. Biol. Assoc. UK* 64, 603–623.
- Barata, C., Calbet, A., Saiz, E., Ortiz, L., Bayona, J.M., 2005. Predicting single and mixture toxicity of petrogenic polycyclic aromatic hydrocarbons to the copepod *Oithona davisae*. *Environ. Toxicol. Chem.* 24, 2992–2999.
- Beaugrand, G., Reid, P.C., Ibañez, F., Lindley, J.A., Edwards, M., 2002. Reorganization of North Atlantic marine copepod diversity and climate. *Science* 296, 1692–1694.

- Berglund, J., Müren, U., Bamstedt, U., Andersson, A., 2007. Efficiency of a phytoplankton-based and a bacterial-based food web in a pelagic marine system. *Limnol. Oceanogr.* 52, 121–131.
- Bhau, M., Cazaux, C., 1987. Description and identification of polychaete larvae. *Oceanis* 13, 595–753.
- Bouley, P., Kimmerer, W.J., 2006. Ecology of a highly abundant, introduced cyclopoid copepod in a temperate estuary. *Mar. Ecol. Prog. Ser.* 324, 219–228.
- Boxshall, G.A., Hasley, S.H., 2004. *An Introduction to Copepod Diversity*. Ray Society, London.
- Bray, R.J., Curtis, J.T., 1957. An ordination of the upland forest communities of southern Wisconsin. *Ecol. Monogr.* 27, 325–349.
- Carter, K.L., Reader, J.P., 2009. Patterns of drift and power plant station entrainment of 0+ fish in the River Trent, England. *Fish. Manag. Ecol.* 5, 447–464.
- Castellani, C., Robinson, C., Smith, T., Lampitt, R.S., 2005. Temperature affects respiration rate of *Oithona similis*. *Mar. Ecol. Prog. Ser.* 285, 129–135.
- Chew, L.L., Chong, V.C., Tanaka, K., Sasekumar, A., 2012. Phytoplankton fuel the energy flow from zooplankton to small nekton in turbid mangrove waters. *Mar. Ecol. Prog. Ser.* 469, 7–24.
- Chihara, M., Murano, M., 1997. *An Illustrated Guide to Marine Plankton in Japan*. Tokai University Press, Tokyo.
- Choi, K.-Y., Kim, Y.-O., Lee, J.-B., Wang, S.-Y., Lee, M.-W., Lee, P.-G., Ahn, D.-S., Hong, J.S., Soh, H.-Y., 2012. Thermal impacts of a coal power plant on the plankton in an open coastal water environment. *J. Mar. Sci. Technol. Taiwan* 20, 187–194.
- Chong, V.C., 1975. Some Aspects of the Biology of the Grey Mullet *Liza malinoptera* Valenciennes (Mugilidae) (B.Sc. thesis) University of Malaya, Malaysia.
- Chong, V.C., 1979. The Biology of White Prawn *Penaeus merguensis* De Man (Crustacea: Penaeidae) in the Pulau Angsa-Klang Strait Waters, (Straits of Malacca) (M.Sc. thesis) University of Malaya, Malaysia.
- Chong, V.C., Sasekumar, A., Wolanski, E., 1996. The role of mangroves in retaining penaeid prawn larvae in Klang Strait, Malaysia. *Mangrove Salt Marshes* 1, 11–22.
- Chuang, Y.-I., Yang, H.-H., Lin, H.-J., 2009. The effects of a thermal discharge from a nuclear power plant on phytoplankton and periphyton in subtropical coastal waters. *J. Sea Res.* 61, 197–205.
- Conversi, A., Peluso, T., Fonda-Umani, S., 2009. Gulf of Trieste: a changing ecosystem. *J. Geophys. Res.* 114, C03S90.
- Coulter, D.P., Sepúlveda, M.S., Troy, C.D., Höök, T.O., 2014. Thermal habitat quality of aquatic organisms near power plant discharges: potential exacerbating effects of climate warming. *Fish. Manag. Ecol.* 21, 196–210.
- Cripps, G., Lindeque, P., Flynn, K.J., 2014a. Have we been underestimating the effects of ocean acidification in zooplankton? *Glob. Chang. Biol.* 20, 3377–3385.
- Cripps, G., Lindeque, P., Flynn, K.J., 2014b. Parental exposure to elevated pCO₂ influences the reproductive success of copepods. *J. Plankton Res.* 0, 1–10.
- Dam, H.G., 2013. Evolutionary adaptation of marine zooplankton to global change. *Ann. Rev. Mar. Sci.* 5, 349–370.
- Edwards, M., Richardson, A.J., 2004. The impact of climate change on the phenology of the plankton community and trophic mismatch. *Nature* 430, 881–884.
- Ferry-Graham, L., Dorin, M., Lin, P., 2008. Understanding Entrainment at Coastal Power Plants: Informing a Program to Study Impacts and Their Reduction. A consultant report for California Energy Commission, Moss Landing Marine Laboratories, USA (38 pp.).
- Fong, C.H., 1978. Studies on *Dorosoma chacunda* (Bleeker) and *Haplochilus melastigma* (McClelland) in a Malaysian Mangrove Estuary (B.Sc. Thesis) University of Malaya, Malaysia.
- Greenwood, M.F.D., 2008. Fish mortality by impingement on the cooling-water intake screens of Britain's largest direct-cooled power station. *Mar. Pollut. Bull.* 56, 723–739.
- Guttikunda, S.K., Jawahar, P., 2014. Atmospheric emissions and pollution from the coal-fired thermal power plants in India. *Atmos. Environ.* 92, 449–460.
- HACH, 1997. *Water Analysis Handbook*. 3rd ed. HACH company, U.S.A.
- Hay, S., 2006. Marine ecology: gelatinous bells may ring change in marine ecosystems. *Curr. Biol.* 16, R679–R682.
- Henderson, P.A., 2015. Are coastal power stations affecting Northern European inshore fish populations? Pisces conservation: power plant ecology: effects of coastal power stations Available online at <http://consult.pisces-conservation.com/2-paper001.html>.
- Henderson, P.A., Seaby, R.M.H., Somes, J.R., 2011. Community level response to climate change: the long-term study of the fish and crustacean community of the Bristol Channel. *J. Exp. Mar. Biol. Ecol.* 400, 78–89.
- Henschke, N., Everett, J.D., Doblin, M.A., Pitt, K.A., Richardson, A.J., Suthers, I.M., 2014. Demography and interannual variability of salp swarms (*Thalia democratica*). *Mar. Biol.* 161, 149–163.
- Highfield, J.M., Eloire, D., Conway, D.V.P., Lindeque, P.K., Attrill, M.J., Somerfield, P.J., 2010. Seasonal dynamics of meroplankton assemblages at station L4. *J. Plankton Res.* 32, 681–691.
- Hirayama, K., Hirano, R., 1970. Influences of high temperature and residual chlorine on marine phytoplankton. *Mar. Biol.* 7, 205–213.
- Hoffmeyer, M.S., Biancalana, F., Barasategui, A., 2005. Impact of a power plant cooling system on copepod and meroplankton survival (Bahia Blanca estuary, Argentina). *Iheringia, Sér. Zool.* 95 (3), 311–318.
- Hooff, R.C., Peterson, W.T., 2006. Copepod biodiversity as an indicator of changes in ocean and climate conditions of the northern California current ecosystem. *Limnol. Oceanogr.* 51, 2607–2620.
- Hutchins, D.A., Fu, F.-X., Zhang, Y., Warner, M.E., Feng, Y., Portune, K., Bernhardt, P.W., Mulholland, M.R., 2007. CO₂ control of *Trichodesmium* N₂ fixation, photosynthesis, growth rates, and elemental ratios: implications for past, present, and future ocean biogeochemistry. *Limnol. Oceanogr.* 52, 1293–1304.
- Intxaurti, L., Villate, F., Uriarte, I., Iriarte, A., Amezttoy, I., 2012. Size-related response of zooplankton to hydroclimatic variability and water-quality in an organically polluted estuary of the Basque coast (Bay of Biscay). *J. Mar. Syst.* 94, 87–96.
- IPCC, 2014. In: Team, C.W., Pachauri, R.K., Meyer, L.A. (Eds.), *Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. IPCC, Geneva, Switzerland, p. 151.
- Ishii, H., Tanaka, F., 2001. Food and feeding of *Aurelia aurita* in Tokyo Bay with an analysis of stomach contents and a measurement of digestion times. *Hydrobiologia* 451, 311–320.
- Jiang, Z.B., Zeng, J.N., Chen, Q.Z., Huang, Y.J., Liao, Y.B., Xu, X.Q., Zheng, P., 2009. Potential impact of rising seawater temperature on copepods due to coastal power plants in subtropical areas. *J. Exp. Mar. Biol. Ecol.* 368, 196–201.
- Kimmel, D.G., Boynton, W.R., Roman, M.R., 2012. Long-term decline in the calanoid copepod *Acartia tonsa* in central Chesapeake Bay, USA: an indirect effect of eutrophication? *Estuar. Coast. Shelf Sci.* 101, 76–85.
- Koski, M., Möller, E.E., Maar, M., Visser, A.W., 2007. The fate of discarded appendicularian houses: degradation by the copepod, *Microsetella norvegica*, and other agents. *J. Plankton Res.* 29, 641–654.
- Krleva, E., Karamfilov, V., Hibaum, G., 2012. Determination of PAH in the black sea water by GC/MS following preconcentration with solid-phase extraction. *Ecol. Chem. Eng. S* 19, 393–403.
- Lardicci, C., Rossi, F., Maltagliati, F., 1999. Detection of thermal pollution: variability of benthic communities at two different spatial scales in an area influenced by a coastal power station. *Mar. Pollut. Bull.* 38, 296–303.
- Lee, C.W., Bong, C.W., 2006. Carbon Flux Through Bacteria in a Eutrophic Tropical Environment: Port Klang Waters. In: Wolanski, E. (Ed.), *The Environment in Asia Pacific Harbour*, Chapter 20. Springer, Netherlands, pp. 329–345.
- Lee, C.W., Bong, C.W., 2012. Microbial Ecology of Port Klang Waters. In: Sasekumar, A., Chong, V.C. (Eds.), *Mangrove and Coastal Environment of Selangor, Malaysia Monograph Series* 14. Institute of Ocean and Earth Sciences, University of Malaya, pp. 163–173.
- Legendre, P., Legendre, L., 1998. *Numerical Ecology*. Elsevier, Amsterdam.
- Lewis, R.B., Seeger, G., 2000. Entrainment and impingement studies at two power plants on Wabash River in Indiana. *Environ. Sci. Pol.* S303–S312.
- Li, W.K.W., McLaughlin, F.A., Lovejoy, C., Carmack, E.C., 2009. Smallest algae thrive as the Arctic ocean freshens. *Science* 326, 539.
- Licandro, P., Ibanez, F., Etienne, M., 2006. Long-term fluctuations (1974–1999) of the salps *Thalia democratica* and *Salpa fusiformis* in the northwestern Mediterranean Sea: relationships with hydroclimatic variability. *Limnol. Oceanogr.* 51, 1832–1848.
- Mackas, D.L., Greve, W., Edwards, M., Richardson, A., Chiba, S., Tadokoro, K., Eloire, D., Mazzocchi, M.G., Batten, S., Head, E., Conversi, A., Fonda-Umani, S., 2012. Changing zooplankton seasonality in a changing ocean: comparing time series of zooplankton phenology. *Prog. Oceanogr.* 97–100, 31–62.
- McConville, K., Halsband, C., Fileman, E.S., Somerfield, P.J., Findlay, H.S., Spicer, J.L., 2013. Effects of elevated CO₂ on the reproduction of two calanoid copepods. *Mar. Pollut. Bull.* 73, 428–434.
- McDonald, M.R., McClintock, J.B., Amsler, C.D., Rittschof, D., Angus, R.A., Orihuela, B., Lutostanski, K., 2009. Effects of ocean acidification over the life history of the barnacle *Amphibalanus amphitrite*. *Mar. Ecol. Prog. Ser.* 385, 179–187.
- Möllmann, C., Müller-Karulis, B., Kornilovs, G., St John, M.A., 2008. Effects of climate and overfishing on zooplankton dynamics and ecosystem structure: regime shifts, trophic cascade, and feedback loops in a simple ecosystem. *ICES J. Mar. Sci.* 65, 302–310.
- O'Connor, M.L., Piehler, M.F., Leech, D.M., Anton, A., Bruno, J.F., 2009. Warming and resource availability shift food web structure and metabolism. *PLoS Biol.* 7, e1000178.
- Parsons, T.R., Maita, Y., Lalli, C., 1984. *Manual of Chemical and Biological Methods for Seawater Analysis*. Pergamon Press, Oxford, p. 173.
- Pielou, E.C., 1969. *An Introduction to Mathematical Ecology*. Wiley, New York, p. 286.
- Pokale, W.K., 2012. Effects of thermal power plant on environment. *Sci. Rev. Chem. Commun.* 2 (3), 212–215.
- Poornima, E.H., Rajadurai, M., Rao, T.S., Anupkumar, B., Rajamohan, R., Narasimhan, S.V., Rao, V.N.R., Venugopalan, V.P., 2005. Impact of thermal discharge from a tropical coastal power plant on phytoplankton. *J. Therm. Biol.* 30, 307–316.
- Qiu, J.W., Qian, P.-Y., 1997. Effects of food availability, larval source and culture method on larval development of *Balanus amphitrite amphitrite* Darwin: implications for experimental design. *J. Exp. Mar. Biol. Ecol.* 217, 47–61.
- Rajagopal, S., Venugopalan, V.P., Van der Velde, G., Jenner, H.A., 2003. Tolerance of five species of tropical marine mussels to continuous chlorination. *Mar. Environ. Res.* 55, 277–291.
- Rice, E., Dam, H.G., Steward, G., 2015. Impact of climate change on estuarine zooplankton: surface water warming in Long Island Sound is associated with changes in copepod size and community structure. *Estuar. Coasts* 38, 13–23.
- Richardson, A.J., 2008. In hot water: zooplankton and climate change. *ICES J. Mar. Sci.* 65, 279–295.
- Saiz, E., Movilla, J., Yebra, L., Barata, C., Calbet, A., 2009. Lethal and sublethal effects of naphthalene and 1, 2-dimethylnaphthalene on naupliar and adult stages of the marine cyclopoid copepod *Oithona davisae*. *Environ. Pollut.* 157, 1219–1226.
- Sany, T.S.B., Salleh, A., Sulaiman, A.H., Sasekumar, A., Rezayi, M., Monazami Tehrani, G., 2013. Heavy metal contamination in water and sediment of the Port Klang coastal area, Selangor, Malaysia. *Environ. Earth Sci.* 69, 2013–2025.
- Sarmiento, H., Montoya, J.M., Vázquez-Domínguez, E., Vaqué, D., Gasol, J.M., 2010. Warming effects on marine microbial food web processes: how far can we go when it comes to predictions? *Philos. Trans. R. Soc. B* 365, 2137–2149.
- Sato, R., Ishibashi, Y., Tanaka, Y., Ishimaru, T., Dagg, M., 2008. Productivity and grazing impact of *Oikopleura dioica* (Tunicata, Appendicularia) in Tokyo Bay. *J. Plankton Res.* 30, 299–309.
- Shannon, C.E., 1948. A mathematical theory of communication. *Bell Syst. Technol. J.* 27, 379–423.
- Sheridan, J.A., Bickford, D., 2011. Shrinking body size as an ecological response to climate change. *Nat. Clim. Chang.* 1, 401–406.

- Shindell, D., Faluvegi, G., 2010. The net climate impact of coal-fired power plant emissions. *Atmos. Chem. Phys.* 10, 3247–3260.
- Suikkanen, S., Pulina, S., Engström-Öst, J., Lehtiniemi, M., Lehtinen, S., Brutemark, A., 2013. Climate change and eutrophication induced shifts in northern summer plankton communities. *PLoS One* 8, e66475.
- Svensen, C., Kjørboe, T., 2000. Remote prey detection in *Oithona similis*: hydromechanical vs. chemical cues. *J. Plankton Res.* 22, 1155–1166.
- Svensen, C., Seuthe, L., Vasilyeva, Y., Pasternak, A., Hansen, E., 2011. Zooplankton distribution across Fram Strait in autumn: are small copepods and protozooplankton important? *Prog. Oceanogr.* 91, 534–544.
- Tan, K.F., 1976. Some Aspects of the Biology of the *Acetes erythraeus* (B.Sc. Thesis) University of Malaya, Malaysia.
- Taylor, C.J.L., 2006. The effects of biological fouling control at coastal and estuarine power plants. *Mar. Pollut. Bull.* 53, 30–48.
- Ter Braak, C.J.F., Smilauer, P., 2002. CANOCO Reference Manual and CanoDraw for Windows User's Guide: Software for Canonical Community Ordination (Version 4.5). Microcomputer Power, Ithaca, New York, p. 500.
- Troedsson, C., Bouquet, J.M., Lobon, C., Novac, A., Nejstgaard, J.C., Dupont, S., Bosak, S., Jakobsen, H.H., Romanova, N., Pankoke, L.M., Isal, A., Dutz, J., Sazhin, A., Thompson, E.M., 2013. Effects of ocean acidification, temperature and nutrient regimes on the appendicularian, *Oikopleura dioica*: a mesocosm study. *Mar. Biol.* 160, 2175–2187.
- Underwood, A.J., 1992. Beyond BACI: the detection of environmental impacts on populations in the real, but variable world. *J. Exp. Mar. Biol. Ecol.* 161, 145–178.
- Underwood, A.J., Chapman, M.G., 1997. Subtidal assemblages on rocky reefs at a cliff-face sewage outfall (North Head, Sydney, Australia): what happened when the outfall was turned off? *Mar. Pollut. Bull.* 33, 7–12.
- Uye, S., 1994. Replacement of large copepods by small ones with eutrophication of embayments: cause and consequence. *Hydrobiologia* 292 (293), 513–519.
- Ward, P., Hirst, A.G., 2007. *Oithona similis* in a high latitude ecosystem: abundance, distribution and temperature limitation of fecundity rates in a sac spawning copepod. *Mar. Biol.* 151, 1099–1110.
- Wiltshire, K.H., Manly, B.F.J., 2004. The warming trend at Helgoland Roads, North Sea: phytoplankton response. *Helgolander Mar. Res.* 58, 269–273.
- Winder, M., Jassby, A.D., 2011. Shifts in zooplankton community structure: implications for food web processes in the upper San Francisco Estuary. *Estuar. Coasts* 34, 675–690.
- Yamada, Y., Ikeda, T., 1999. Acute toxicity of lowered pH to some oceanic zooplankton. *Plankton Biol. Ecol.* 46, 62–67.
- Young, C.M., Sewell, M.A., Rice, M.E., 2002. Atlas of Marine Invertebrate Larvae. Academic Press, California.
- Zamora-Terol, S., McKinnon, A.D., Saiz, E., 2014. Feeding and egg production of *Oithona* spp. in tropical waters of North Queensland, Australia. *J. Plankton Res.* 36, 1047–1059.