

Does the copepod community of the Ganges River estuary of India indicate ecological stress?

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Abstract

Copepods are considered as indicators of ecological stresses of estuaries. Spatial-temporal changes of diversity, abundance, and dominance of the copepod community of India's Ganges River estuary were hypothesised to result from abiotic fluctuations, seasonal change and the COVID-19 lockdown which possibly affected the habitat. This hypothesis was studied by sampling the estuary on different occasions between February 2019 and October 2020. Salinity and pH showed significant temporal variability but not significant spatial variability. Water temperature showed no significant spatial-temporal variability. Salinity showed significant negative relationships with diversity and abundance of the copepod community. After the COVID-19 lockdown (between unlock phase 1 to 5), nutrient status of the estuary did not show significant temporal variation. The copepod community mostly comprised calanoids except for the cyclopoids *Oithona* spp. Relative abundances of *Pseudodiaptomus binghami*, *Eucalanus crassus* and *Labidocera euchaeta* declined but *Paracalanus indicus*, *Acrocalanus gibber*, *Acrocalanus gracilis*, *Acartia spinicauda*, *Acartia sewelli* and *Oithona brevicornis* increased after COVID-19 lockdown. *Bestiolina similis* was the most abundant copepod and along with *Acartiella tortaniformis*, *Acartia spinicauda*, *Paracalanus parvus* and *O. brevicornis* co-dominated the community on various occasions. Spatial ordination of the copepod community was not significant either in the pre- or post-lockdown period. Diversity and total abundance of the copepod community were affected neither by seasonal change nor by temporal variability of water temperature and pH. Temporal variability of species richness was significant only in the post-lockdown period. Pre- vs. post-lockdown comparisons of diversity indices and total abundance of the copepod community were not significant. Overall, copepods were not much affected by abiotic and seasonal changes, and the COVID-19 lockdown. That indicates their flexibility to habitat variability and less suitability as indicators of chronic ecological stresses of excessively large river-estuaries.

1. Introduction

A river-estuary is a transitional ecosystem between a sea and a river (Whitfield, 1992). Large river-estuaries of the world are often depleted and degraded ecosystems because numerous human-interventions have modified their natural characteristics (Lotze et al., 2002; Harrison et al., 2008). Worldwide, large river-estuaries are ecologically stressed from many human-interventions (e.g. agricultural, industrial, traffic, fishing, pollution, water abstraction), and those stresses are often inseparable from natural stresses such as from abiotic changes and seasonal variability (Elliot and Whitfield, 2011; Chen et al., 2013; Feebarani et al., 2016; Mitra et al., 2018; Barletta et al., 2019; Mitra et al., 2019). Ecological communities of those river-estuaries are often living under stress so many of them have become resilient to abiotic changes, seasonal vulnerability and human-interventions; however, many others are failing to cope with the increasing stress levels of their habitats and are becoming severely degraded (Millward and Grant, 1995; Araujo et al., 2017).

Tropical river-estuaries of Brazil, Australia, Malaysia and India support a wide diversity of mesozooplankton, mostly copepods (Costa et al., 2008; Duggan et al., 2008; Chew and Chong, 2011; Bhattacharya et al., 2015). The distribution of copepods in tropical river-estuaries is largely determined by tidal movements; however, gradients of salinity, water temperature, pH, nutrient status and many forms of human intervention (e.g. pollution, damming) also influence their distribution within an estuary (Sinha et al., 1996; Araujo et al., 2008; Costa et al., 2008; Chew et al., 2015). In tropical estuaries, seasonal variability limits mesozooplankton diversity and distribution (Costa et al., 2008; Chew and Chong, 2011; Bhattacharya et al., 2015). Nutrient availability of tropical river-estuaries increases in the late monsoon (wet season) due to the large input of catchment runoff, and the

responding phytoplankton bloom in the early post-monsoon in turn supports a diverse copepod community (Sarkar et al., 1986; Madhu et al., 2007; Biswas et al., 2010; Bhattacharya et al., 2015; Patil and Anil, 2019).

Plankton communities of the Ganges River estuary (GRE) of India are largely influenced by semi-diurnal tides and seasonal changes, especially the arrival and departure of the south-west monsoon season which annually brings about 1800 mm of rain to the Indian Sundarban region (Mukhopadhyay et al., 2006; Choudhury et al., 2015; Bhattacharya et al., 2015). The Muriganga is a relatively small bifurcation of the GRE that shares all the common physical-chemical properties of the main channel (Mukhopadhyay et al., 2006). Nutrient status of the Muriganga plays a role in regulating temporal changes of the phytoplankton community, but its impact on the copepod community has yet to be studied (Mukhopadhyay et al., 2006; Choudhury et al., 2015). The Muriganga section receives household sewage and runoff from agriculture and aquaculture farms which contribute to bioaccumulation of heavy metals in the flora and fauna of local estuaries (Bhattacharya et al., 1994). Hourly variability of water temperature and pH are considerably less in the Muriganga, so hardly limit the diversity and distribution of copepods (Paul et al., 2019). *Bestiolina similis* generally dominates the copepod community of Muriganga; however, *Paracalanus parvus* is also abundant and at times co-dominates the estuary with *B. similis* (Paul et al., 2019). If the Muriganga is perturbed by natural disturbance (e.g. cyclone), the copepods *Acartiella tortaniformis* and *Acartia spinicauda* become highly abundant for a few weeks to months and co-dominate with *B. similis* and *p. parvus* (Paul et al., 2020a). The copepod community of the Muriganga mostly comprises estuarine residents, but also includes occasional marine intruders such as *Acrocalanus gibber* and *Eucalanus elongatus*; however, their abundances remain generally low (Sarkar et al., 1986; Paul et al., 2019). Copepods of the Muriganga possibly experience varying degrees of ecological stress, some of which are natural (e.g. tidal, seasonal and abiotic changes) while others are human-mediated (e.g. pollution) (Sinha et al., 1996; Bhattacharya et al., 2014b; Bhattacharya et al., 2015; Paul et al., 2019, 2020a).

On 30 January 2020, India reported its first case of the COVID-19 and from 24 March to 31 May 2020 (68 days) the country was in complete lockdown affecting all sorts of activities (economic, industrial, agricultural, transport, fishing) and mobility of people (Mishra et al., 2020). India's unlock phase 1 started on 1 June 2020. Mondal (2020) suggested that the lockdown period is a rare opportunity to revive aquatic ecosystems of India, including the Hooghly-Matla estuarine complex which are two different sections of GRE. After the COVID-19 lockdown, many economic, industrial, agricultural, transport and fishing activities have opened phase by phase. The COVID-19 lockdown reduced the input of nutrients to coastal waters of India which in consequence negatively affected the primary production of coastal lagoons such as the Chilika Lagoon (Mishra et al., 2020). The COVID-19 pandemic and related lockdown were unforeseen in early 2019 when the current study was conceptualized. So, data related to human-interventions on the Muriganga and their influence on copepod diversity and distribution were not collected. No study was permitted during lockdown so any changes of habitat and copepod community cannot be estimated accurately for those days. Copepod sampling resumed after lockdown was partially lifted (i.e. start of unlock 1) in June 2020. Human impacts (except domestic sewage dumping) on the Muriganga were less in 2020 during the COVID-19 lockdown and unlock phases which restricted various economic and development activities that daily affect the estuary (see details in the study site). Objective in this study was to evaluate the effects of abiotic variability, seasonal change and the COVID-19 lockdown (assumed as a period of minimum human intervention) on the copepod community of the Muriganga. It was hypothesised that spatial-temporal changes of diversity, abundance, and dominance within the copepod

community were the result of changes such as abiotic fluctuations, seasonal change and the COVID-19 lockdown which possibly affected the Muriganga section of the GRE between 2019 and 2020.

2. Methods

2.1. *Study site*

The Muriganga section of the GRE (Fig. 1) has moderately developed mangrove vegetation and an intensely cultivated hinterland. This stretch of the GRE experiences three seasons per year, which is typical for the Indian Sundarban: pre-monsoon (March to May), monsoon (15 June to 15 September and seldom extended) and post-monsoon (October to February) (Bhattacharya et al. 2015). The Muriganga generally remains mesohaline and slightly basic (pH > 8) except during monsoon months when it receives excessive rainfall (Mukhopadhyay et al., 2006; Choudhury et al., 2015). Commercial and subsistence fishing activities are routine in the Muriganga and generally increase in the monsoon, ferry service and cargo ships (often dump burnt diesel in the river) are also frequent in this section of Ganges River. The study sites are not directly influenced by inputs of industrial effluents; however, the upstream stretches of the Muriganga (between Diamond Harbour and Namkhana) have considerable industrial activities such as dry-dock, lead and battery factories, and many bricks kilns. The Diamond Harbour to Sagar (including Muriganga) stretch of the GRE receives high loads of nitrates, phosphates and heavy-metals (Mitra et al., 2018; Mitra et al., 2019). The Muriganga does receive agricultural runoff from its immediate surroundings because there are many agricultural fields and aquaculture farms in the close vicinity. On a daily basis, 177 trawlers/fishing boats, 113 passenger boats and 69 industrial ships operate in the Muriganga stretch. Such industrial, agricultural, fishing and travel activities on the Muriganga and its catchment were completely banned during the COVID-19 lockdown and after the COVID-19 lockdown those activities restarted on a phased basis between unlock 1 to unlock 5.

At three permanent sampling sites S1, S2, and S3 on the Muriganga, copepods (size class > 200 μm) were sampled on various occasions from February 2019 onwards. The sites were 500 m apart: S1 (21°44'53.8"N, 88°12'46.2" E) towards the upper end of the estuary, S3 (21°44'55.4" N, 88°12'36.8" E) towards the mouth, and S2 (21°44'55.7"N, 88°12'40.0"E) equidistant between S1 and S3 (Fig. 1). The GRE is an excessively large estuary and even its Muriganga section is fairly large (about 30 K.M.). Budgetary limitations restricted the authors from any significant spatial coverage of the Muriganga section so S1, S2 and S3 sites shall be treated more like replicates rather than independent sampling stations. All sites were sampled from a motor boat at night on high tide. Copepod samples were collected between February 2019 and October 2020 on various occasions to fulfil the objective of seasonal sampling; however, maintaining any specific sampling interval (e.g. monthly / bi-monthly etc.) was not possible for limited sampling permissions granted by the concerned authorities, budgetary constraints, less availability of boat during fishing seasons. For the COVID-19 restrictions, sampling had to be stopped between January and May 2020. Between June and October 2020, all sites were revisited multiple times. Samples collected during 2019 were assumed to represent pre-lockdown conditions and those collected from June 2020 onwards were assumed to represent the post-lockdown condition (i.e. unlock 1 to 5).

2.2. *Measurement of abiotic data and nutrients*

On all the occasions of copepod sampling (before and after the COVID-19 lockdown), salinity, water temperature ($^{\circ}\text{C}$), and pH were measured from 0.5–1 m water depth using a hand-held multi-parameter probe (YSI-1030,

USA). In 2019, there was no plan of assessing the nutrient status of the Muriganga so no data were collected. For that reason pre- vs. post-lockdown nutrient status comparison was not possible. Following the COVID-19 lockdown, nutrient status was measured to provide a post-lockdown reference of nutrient status of the estuary for any future work on water quality of the Muriganga. From June 2020, on each occasion of copepod sampling, a water sample was collected from 0.5–1 m water depth on high tide (additional water samples were also collected on occasions when copepod sampling was not conducted). On each occasion one sampling site was randomly picked between S1, S2 and S3 because Paul et al. (2019) had shown that there was no significant spatial variation of nutrient status between S1, S2 and S3. Samples were collected in 50 ml plastic falcon tubes for analysis of Total Nitrogen (TN), Nitrate-nitrogen (NO₃-N), Dissolved Inorganic Phosphate (DiP) and Dissolved Organic Phosphate (DOP). All water samples were filtered on board through GF/F filter papers (1.2 µm pore size) and the filtered samples transported to the laboratory within 24 hours in ice. Within 48 hours of transport, the filtered water samples were outsourced to the Good Earth laboratory (specialists in water quality of the Indian Sundarban region) for the measurements of the concentrations (µM) of TN, NO₃-N, DiP and DOP. At the Good Earth laboratory, nutrients were measured using a 4200UV/Visible spectrophotometer (Aqua Quest, Cecil, U.K) following the procedures of Grasshoff et al. (2009).

2.3. Copepod collection and identification

On each occasion of copepod sampling, 100 L of estuarine water was collected through a 10 L plastic bucket from 0.5–1 m water depth and filtered through a conical plankton net (60 cm diameter and 150 cm in length, mesh size of 200 µm). Each filtrate was decanted into a sample bottle and 5 ml of 4% buffered formalin was added to preserve the copepod sample. Samples were transported to the laboratory within 24 hours of collection. In the laboratory, multiple aliquot samples each of 1 ml were taken from the filtrate and each aliquot was placed on a Sedgewick Rafter counting cell and examined under a stereo-microscope (Bestscope-BS30T, China). Copepods were identified to species following the taxonomic literature of Kasturirangan (1963) and the abundance of each species was expressed as individuals per cubic metre (i.e. ind.m⁻³).

2.4. Statistical analysis

All the analyses were performed using CRAN-R3.6.3 (R Core Team, 2020). Results of statistical tests were presented with corresponding t, W, F, K-W chi-square and p values and degrees of freedom (DF).

2.4.1. Analysis of abiotic and nutrient data

Salinity, water temperature and pH data showed non-parametric distributions (checked using Shapiro-Wilk tests), so their spatial variations (if any between S1, S2 and S3) were assessed by Kruskal-Wallis tests and their temporal variations were evaluated through multiple generalised linear model (GLM) of log-normal family. Salinity, water temperature and pH values of the pre- and post-lockdown periods were not compared because that was outside the aim of the study.

TN and DOP data followed parametric distributions so their temporal variations were assessed by multiple GLM of Gaussian family considering TN and DOP as the response variables of the regression equations; however, NO₃-N and DiP data showed non-parametric distributions so their temporal changes were tested through multiple GLM of log-normal family, considering NO₃-N and DiP as the response variables of the regression equations.

2.4.2. Analysis of copepod data

Each time a copepod assemblage was sampled, corresponding values of species richness, Shannon-index (H) (Shannon and Weaver, 1963) and Pielou's index (J) (Pielou, 1966) were calculated by using the 'Vegan' package version 2.5.6 (Oksanen et al., 2019). Site-specific relative abundances of copepods were calculated. Dominant and or co-dominant status of the few frequently sampled copepods was calculated following the index of dominance formula adopted by Bhattacharya et al. (2014a) i.e. $Y_i = (N_i/N) * F_i$ where Y_i is the dominance of species i , N_i is the number of individuals of species i at all sites (i.e. S1, S2 and S3), N is the number of all species at all sites, and F_i is the frequency of sites at which species i occurs. Species with a Y_i value greater than 0.02 were considered dominant species of the habitat (for details see Bhattacharya et al. (2014a).

Datasets of 2019 and 2020 were combined for analysing seasonal changes of species richness, Shannon-index (H), Pielou's index (J), and total abundance of copepods. Values of H index and J index were normally distributed, but the values of species richness and total abundance followed non-parametric distributions. Seasonal variability of H and J indices was evaluated through one-way ANOVA, but seasonal changes of species richness and total abundance were evaluated using Kruskal-Wallis tests. Multiple GLM (including interactive models of multiple parameters) was used for evaluating the impacts of temporal variability of salinity, water temperature and pH levels on the copepod community. In case the response variable was either H index or J index, a GLM of Gaussian family was used because the response variable was normally distributed. In case the response variable was either species richness or total abundance of copepods, then a GLM of log-normal family was used because the response variable followed non-parametric distribution.

Spatial variations (between S1, S2 and S3) of the total copepod abundance, species richness, H and J indices were evaluated separately for the pre- and post-lockdown periods by conducting either Kruskal-Wallis tests or one-way ANOVA. Temporal variations of species richness, total abundance, H index and J index were assessed separately for the pre- and post-lockdown periods by conducting multiple GLM of Gaussian family (for H index and J index which were normally distributed) or log-normal family (for species richness and total abundance which followed non-parametric distributions). Pre- vs. post-lockdown comparisons of species richness, total abundance, H and J indices were conducted through t-tests or Mann-Whitney U-tests depending on the normality of the response variable. For the pre- and post-lockdown periods, separate ordination analysis of the copepod assemblages (abundance data was square-root transformed) was done through the Non-metric Multidimensional Scaling (NMDS) using the Bray–Curtis measure of dissimilarity ('Vegan' package version 2.5.6). Inference of dimensionality (two dimensions were considered) was taken after examining the stress-scores of NMDS and the Shepard diagrams which were drawn through the 'stressplot' function and non-metric as well as linear fit R^2 scores were extracted. For drawing NMDS biplots, 'ggplot2' package version 3.3.2 (Wickham et al., 2016) was used. For the pre- and post-lockdown periods, copepod assemblages at S1, S2 and S3 were compared (i.e. centroids) using PERMANOVA (Adonis test, permutations = 999, method = Bray-Curtis, package: 'Vegan' version: 2.5.6). In this regard the assumption of homogeneity of multi-variate dispersion was through ANOVA.

3. Results

3.1. Abiotic variability of Muriganga

Spatial variations (i.e. between S1, S2 and S3) of salinity, temperature and pH are summarized in Table 2. In the pre-lockdown period, spatial variation of salinity was not significant (K-W Chi-Square: 0.08, DF = 2, P = 0.96), nor was it for water temperature (K-W Chi-Square: 0.4, DF = 2, P = 0.8) and pH (K-W Chi-Square: 0.14, DF = 2, P = 0.9) levels. The highest salinity (19.3) was recorded in May and the lowest in August (8.60), with salinity increasing from November onwards (Fig. 2a, Table 2). Temporal change of salinity was significant ($t = -2.52$, DF = 14, P = 0.02). The highest water temperature (31.30°C) was recorded in May, Water temperature subsequently declined with the lowest water temperature (20.40°C) measured in December (Fig. 2a, Table 2); however, temporal variation of water temperature was not significant ($t = -1.67$, DF = 14, P = 0.12). An alkaline pH of the Muriganga was evident on most sampling occasions. The highest pH (8.64) was recorded in December, and pH values varied significantly with time ($t = 2.98$, DF = 14, P = 0.01) (Fig. 2a, Table 2).

Table 1

Site-specific variations of salinity, temperature and pH levels of Muriganga section of Ganges River estuary before (February-December 2019) and after (June-October 2020) COVID-19 lockdown (24 March to 31 May 2020) of India.

Sampling Interval	Salinity			Water temperature (°C)			pH			
	S1	S2	S3	S1	S2	S3	S1	S2	S3	
Pre-lockdown	Median	12.25	12.60	12.59	23.40	23.30	23.30	8.10	8.11	8.14
	Range	8.90–19.30	8.85–19.20	8.60–19.20	20.40–31.20	20.70–31.10	20.70–31.30	6.10–8.64	6.10–8.60	6.20–8.60
Post-lockdown	Median	16.20	16.30	16.30	29.70	29.90	29.90	8.31	8.22	8.25
	Range	5.10–18.30	5.10–18.20	4.80–19.30	28.30–30.20	28.30–30.80	28.40–30.20	6.00–8.41	7.66–8.56	7.76–8.51

Table 2

Relative abundance (%) of copepod species sampled from three sampling sites (S1, S2, and S3) on the Muriganga section of the Ganges River estuary before (February to December 2019) after (June to October 2020) the COVID-19 lockdown (24 March to 31 May 2020) of India.

Copepod species	S1		S2		S3	
	Pre-lockdown	Post-lockdown	Pre-lockdown	Post-lockdown	Pre-lockdown	Post-lockdown
<i>Paracalanus parvus</i>	12.86	09.62	10.60	08.30	09.53	08.69
<i>P. aculeatus</i>	04.77	03.01	04.79	03.19	04.14	04.09
<i>P. indicus</i>	00.47	03.79	01.18	04.49	01.02	04.09
<i>Parvocalanus dubia</i>	02.03	00.78	02.13	03.70	04.40	03.49
<i>Acrocalanus gibber</i>	00.00	02.26	00.00	00.59	00.00	00.00
<i>A. gracilis</i>	01.69	04.24	02.75	03.77	0.86	02.90
<i>A. longicornis</i>	01.86	01.50	00.74	02.52	00.90	02.36
<i>Bestiolina similis</i>	21.30	20.47	21.73	19.00	19.45	16.60
<i>Parvocalanus crassirostris</i>	03.26	04.42	02.47	01.33	01.53	03.71
<i>Acartia spinicauda</i>	08.70	13.84	09.83	13.34	10.31	12.59
<i>Acartiella sewelli</i>	00.00	02.37	00.00	02.65	00.00	02.32
<i>Acartia tonsa</i>	02.33	00.00	01.40	00.61	01.17	01.80
<i>A. tropica</i>	04.59	00.00	04.56	02.98	06.63	03.41
<i>Acartiella tortaniformis</i>	16.65	17.50	11.31	14.47	13.13	11.59
<i>Pseudodiaptomus binghami</i>	02.50	01.41	02.49	00.59	02.90	00.56
<i>P. serricaudatus</i>	02.96	03.64	06.84	01.22	08.87	04.65
<i>Canthocalanus pauper</i>	00.00	01.41	02.10	02.39	02.55	01.66
<i>Eucalanus subcrassus</i>	00.47	00.82	03.03	01.80	01.15	01.69
<i>E. crassus</i>	01.55	00.00	01.41	00.00	00.86	00.58
<i>Labidocera euchaeta</i>	01.08	00.00	01.40	00.00	00.86	00.00
<i>L. acuta</i>	01.57	00.72	01.76	01.80	01.17	00.58
<i>Temora turbinata</i>	00.00	01.41	01.24	00.61	00.76	01.80
<i>Corycaeus danae</i>	01.08	00.82	01.03	03.03	00.86	01.10
<i>Oncaea venusta</i>	03.16	00.00	00.00	00.00	00.00	02.36

	S1		S2		S3	
<i>Oithona similis</i>	00.47	00.00	00.36	00.61	00.61	00.00
<i>O. brevicornis</i>	04.68	05.98	04.85	07.01	06.35	07.39

In the post-lockdown period, salinity (K-W Chi-Square: 0.14, DF = 2, P = 0.93), water temperature (K-W Chi-Square: 0.005, DF = 2, P = 0.99) and pH (K-W Chi-Square: 0.03, DF = 2, P = 0.98) showed no significant spatial variation. The highest salinity (19.3) was recorded in June and the lowest salinity (4.8) in October (Fig. 2b, Table 2). Temporal change of salinity was significant ($t = -6.22$, DF = 14, $P < 0.001$). The highest water temperature (30.2°C) was recorded in October and the lowest (28.3 °C) in June (Fig. 2b, Table 2); however, temporal variation of water temperature was not significant ($t = -1.42$, DF = 14, $P = 0.19$). An alkaline pH of the Muriganga was observed in the post-lockdown period (Fig. 2b, Table 2), with the highest pH (8.56) recorded in June, whereas an acidic pH (6) was evident in October 2020 (Fig. 2b, Table 2). The temporal variation of pH was significant ($t = -3.40$, DF = 14, $P = 0.006$).

3.2. Nutrient status of Muriganga after the COVID-19 lockdown

In the post-lockdown period (from the end of complete lockdown to unlock 5 phase), TN ranged from 68.68 to 87.79 μM (mean = 79.40, SD = 6.15) (Fig. 3a) and $\text{NO}_3\text{-N}$ ranged from 34.22 to 55.14 μM (median = 39.35, SE = 1.12) (Fig. 3a). DiP ranged from 1 to 1.84 μM (median = 1.6, SE = 0.04) (Fig. 3b) and DOP from 1.10 to 1.54 μM (mean = 1.25, SD = 0.12) (Fig. 3b). Temporal variations of TN ($t = -0.08$, DF = 17, $P = 0.9$), $\text{NO}_3\text{-N}$ ($t = 0.96$, DF = 17, $P = 0.35$), DiP ($t = 1.86$, DF = 17, $P = 0.08$) and DOP ($t = -0.99$, DF = 17, $P = 0.33$) levels were not significant.

3.3. Copepod diversity, abundance, dominance and distribution

The copepod community of the Muriganga mostly consisted of calanoid copepods, except for the cyclopoids *Oithona brevicornis* and *O. similis* (Table 2). The highest total abundance of copepods was observed at S3 and the lowest total abundance at S1 (Table 3). Maximum species richness was 19 in the pre-lockdown period and 22 in the post-lockdown period (Table 3). No significant spatial variation of species richness was found in the pre-lockdown (K-W chi-square = 0.49, DF = 2, $P = 0.78$) and post-lockdown (K-W chi-square = 1.13, DF = 2, $P = 0.56$) periods. The total abundance of copepods (Table 3) showed no significant spatial variation in the pre-lockdown (K-W chi-square = 5.36, DF = 2, $P = 0.06$) and post-lockdown (K-W chi-square = 4.26, DF = 2, $P = 0.11$) periods. The maximum Shannon index (H) was 2.89 during pre-lockdown and 2.81 during post-lockdown (Fig. 4ab). The Shannon index (Fig. 4a) showed no significant spatial variation in the pre- ($F = 0.24$, DF = 2, $P = 0.8$) and post-lockdown periods ($F = 1.352$, DF = 2, $P = 0.3$) (Fig. 4a). The maximum value of Pielou's evenness (J) index was 0.98 in the pre-lockdown period (Fig. 4a) and 0.92 in the post-lockdown period (Fig. 4b). The Pielou's index (Fig. 4ab) showed no significant spatial variation either in the pre- ($F = 0.09$, DF = 2, $P = 0.90$) or post-lockdown ($F = 0.81$, DF = 2, $P = 0.47$) periods. The pre- vs. post-lockdown comparisons of total copepod abundance ($W = 114$, DF = 26, $P = 0.26$), species richness ($W = 108$, DF = 26, $P = 0.38$), Shannon index ($t = 0.85$, DF = 26, $P = 0.40$) and Pielou's index ($t = -0.97$, DF = 26, $P = 0.34$) showed no significant variations.

Table 3

Total abundance (ind.m⁻³) and species richness of copepods sampled before (February 2019 to December 2019) and after (June to October 2020) the COVID-19 lockdown (24 March to 31 May 2020) of India from three sampling sites (S1, S2 and S3) on the Muriganga section of the Ganges River estuary.

		S1		S2		S3	
Sampling Period	Sampling Date	Total Abundance	Species Richness	Total Abundance	Species Richness	Total Abundance	Species Richness
	25/02/2019	36100	19	35610	14	35970	13
	14/05/2019	46000	07	67000	13	83000	15
Pre-lockdown	24/08/2019	29166	17	30000	19	30833	16
	18/11/2019	51333	09	74733	12	88000	11
	28/12/2019	13333	08	21667	09	23767	09
	05/06/2020	28000	08	54833	10	55533	11
Post-lockdown	12/06/2020	44433	12	59950	12	68200	16
	09/09/2020	39600	13	63783	17	86667	20
	03/10/2020	72000	17	73600	20	100800	22

Bestiolina similis was the most abundant copepod and maintained its dominant status throughout the study (Table 3, 4). Of the other species recorded, *Acartiella tortaniformis*, *Acartia spinicauda*, *Acartia tropica*, *Paracalanus parvus*, *Paracalanus aculeatus*, *Pseudodiaptomus serricaudatus* and *Oithona brevicornis* were abundant on various sampling occasions of 2019–2020 (Table 3). According to the index of dominance (Table 4), species such as *A. tortaniformis*, *A. spinicauda*, *P. parvus* and *O. brevicornis* co-dominated the estuary on different occasions of the pre-and post-lockdown periods (Table 4). For the pre- and post-lockdown periods, two-dimensional ordination NMDS plots of the copepod community are presented in Figs. 5a and b. For the pre- and post-lockdown datasets, NMDS stress scores were 0.084 and 0.11 respectively, and on each occasion solution was reached in 20 simulations. The Shepard diagram (i.e. observed dissimilarity vs. ordination distance) of the pre-lockdown dataset showed non-metric fit $R^2 = 0.993$ and linear fit $R^2 = 0.962$ and when the Shepard diagram was drawn on the post-lockdown dataset the non-metric fit $R^2 = 0.987$ and linear fit $R^2 = 0.947$. A PERMANOVA test showed no significant difference in the copepod community sampled from S1, S2 and S3 in the pre-lockdown period (DF = 2, Sum of Square = 0.11, Pseudo-F = 0.59, $R^2 = 0.89$, P = 0.77) (the assumption of homogeneity of multivariate dispersion was not violated (ANOVA: DF = 2, F = 0.59, P = 0.57)). In the post-lockdown period too, PERMANOVA demonstrated no significant variation (DF = 2, Sum of Square = 0.09, Pseudo-F = 0.97, $R^2 = 0.17$, P = 0.48) of the copepod community sampled from S1, S2 and S3 (the assumption of homogeneity of multivariate dispersion was also met (ANOVA: DF = 2, F = 0.86, P = 0.45)).

Table 4

Index of dominance of few frequently sampled copepod species of the Muriganga section of the Ganges River estuary before (February to December 2019) and after (June to October 2020) COVID-19 lockdown (24 March to 31 May 2020) of India.

Sampling Period	Sampling Date	Bestiolina similis	Acartiella tortaniformis	Acartia spinicauda	Paracalanus parvas	Oithona brevicornis
	25/02/2019	0.12	0.01	0.06	0.08	0.06
	14/05/2019	0.23	0.18	0.13	0.11	0.04
Pre-lockdown	24/08/2019	0.13	0.06	0.04	0.11	0.02
	18/11/2019	0.22	0.15	0.09	0.11	0.08
	28/12/2019	0.29	0.28	0.11	0.07	0.02
	05/06/2020	0.22	0.14	0.21	0.07	0.08
Post-lockdown	12/06/2020	0.19	0.14	0.12	0.11	0.08
	09/09/2020	0.17	0.14	0.10	0.08	0.07
	03/10/2020	0.16	0.13	0.11	0.07	0.05

Species richness ($t = -1.93$, $DF = 14$, $P = 0.07$) and total abundance of copepods ($t = 0.38$, $DF = 14$, $P = 0.70$) did not show any significant temporal variation during the pre-lockdown period (Fig. 4a); however, in the post-lockdown period species richness ($t = 7.74$, $DF = 11$, $P < 0.001$) and total abundance increased significantly with time ($t = 2.78$, $DF = 11$, $P = 0.01$) (Fig. 4b). Temporal variation of Pielou's index was significant ($t = -2.23$, $DF = 14$, $P = 0.04$) in the pre-lockdown period (Fig. 4a) but not in the post-lockdown ($t = -1.08$, $DF = 11$, $P = 0.30$) (Fig. 4b). At S1, S2 and S3, the relative abundance of calanoid copepods such as *B. similis*, *P. parvus*, *P. aculeatus*, *A. tropica*, *Pseudodiaptomus binghami*, *Eucalanus crassus* and *Labidocera euchaeta* declined in the post-lockdown period, but the relative abundance of *P. indicus*, *Acrocalanus gibber*, *A. gracilis*, *Acartia spinicauda* and *A. sewelli* was higher in the post-lockdown period compared to the pre-lockdown period (Table 3). The relative abundances of *Acartiella tortaniformis* at S1 and S2 were higher in the post-lockdown period compared to those in the pre-lockdown period. At S3 the relative abundance of *A. tortaniformis* declined in the post-lockdown period compared to its relative abundance in the pre-lockdown period (Table 3). The relative abundance of the cyclopoid *Oithona brevicornis* was higher in the post-lockdown period than in the pre-lockdown period (Table 2). The dominance of *Acartia spinicauda* and *O. brevicornis* increased in the post-lockdown period compared to that in the pre-lockdown period (Table 4).

3.4. Effects of seasonal change and abiotic variability on copepods

Species richness did not show any distinct variation (K-W chi-square = 5.35, $DF = 2$, $P = 0.07$) between the pre-monsoon, monsoon and post-monsoon. Total copepod abundance also did not vary significantly between seasons (K-W chi-square = 0.68, $DF = 2$, $P = 0.7$). Neither the Shannon index (H) ($F = 2.37$, $DF = 2$, $P = 0.11$) nor the

Pielou's index (J) ($F = 1.15$, $DF = 2$, $P = 0.33$) showed any significant variation between seasons. Temporal variability of salinity showed significant negative associations with species richness ($t = -3.11$, $DF = 26$, $P = 0.005$), total abundance ($t = -2.99$, $DF = 26$, $P = 0.006$) and Shannon index (H) ($t = -2.44$, $DF = 26$, $P = 0.022$). Pielou's index (J) ($t = -0.02$, $DF = 26$, $P = 0.987$), however, was not significantly associated with the temporal variability of salinity. Salinity while interacting with water temperature and pH levels showed no significant temporal relationships with species richness, total abundance, Shannon index (H) and Pielou's index (J). Neither water temperature nor pH nor their interactions showed significant temporal associations with any diversity index and total abundance of copepods of the Muriganga.

4. Discussion

4.1. *Habitat characteristics*

Spatial variations of sub-surface salinity, water temperature and pH levels of the Muriganga were not significant, results that ratify the works of Paul et al. (2019) and Paul et al. (2020b). The highest salinity was recorded in May 2019, possibly due to the higher rates of surface water evaporation of the Muriganga in the middle of pre-monsoon (Mukhopadhyay et al., 2006). The Muriganga is macrotidal and not far from the mouth of the GRE, so it receives saltwater intrusion on a regular basis from the Bay of Bengal, which causes significant temporal variation of its salinity (Mukhopadhyay et al., 2006; Choudhury et al., 2015). Water temperature did not vary significantly on a temporal scale, a result consistent with Bhattacharya et al. (2015) who observed seasonal temperature of mangrove estuaries to vary only within a narrow range while conducting a multi-year mesozooplankton diversity study of the Indian Sundarban. The pH of the Muriganga was basic on most sampling occasions. Choudhury et al. (2015) observed an alkaline pH profile of the Muriganga throughout the year unless the estuary receives a lot of rain in the monsoon which temporarily depresses the pH profile.

The nutrient cycle of the lower stretch of the Ganges River, including the Muriganga, is largely impacted by non-point sources, such as indiscriminate dumping of domestic sewage, industrial wastewater discharge, agricultural run-off, and run-off of fish and fin-fish farms that are widespread in the Namkhana and Sagar Island regions through which the Muriganga runs towards the sea (Bhattacharya et al., 1994; Mitra et al., 2018; Mitra et al., 2019). The Water Quality Index (WQI) value of the GRE indicates a 'bad' condition of the estuary, which is detrimental for most forms aquatic biodiversity including plankton (Mitra et al., 2018). Yet the nutrient status of the GRE in particular the Muriganga section is not monitored on a regular basis, so delineating baseline levels for TN, $\text{NO}_3\text{-N}$, DiP and DOP for the Muriganga is difficult. After the COVID-19 lockdown ended on 31 May 2020, TN and $\text{NO}_3\text{-N}$ levels of the Muriganga did not vary significantly between unlock phase 1 to 5. The TN and $\text{NO}_3\text{-N}$ levels were much higher compared to the TN and $\text{NO}_3\text{-N}$ levels of river-estuaries less impacted by human interventions (Araujo et al., 2017; Mitra et al., 2018). The nitrate concentration of the Muriganga depends a lot on the turbidity of the estuary and the presence of faecal coliforms, both of which generally increase in the monsoon season between June and September (Mukhopadhyay et al., 2006; Bhattacharya et al., 2015; Mitra et al., 2018). Like many other tropical river-estuaries, the GRE is phosphate limited and the present results support earlier observations (Mukhopadhyay et al., 2006; Araujo et al., 2008; Choudhury et al., 2015; Mitra et al., 2018). In the Muriganga, the DiP levels were always higher than the DOP levels which demonstrates that the estuary receives considerable catchment run-off from agriculture and aquaculture farms (Bhattacharya et al., 1994; Mukhopadhyay et al., 2006; Mitra et al., 2019). During the monsoon (July to September), the Muriganga receives

large inputs of nutrients from catchment run-off and there is spatial heterogeneity of nutrient concentration (Mukhopadhyay et al., 2006; Bhadury and Singh, 2020). In 2020, the monsoon season largely coincided with the unlock phases of the COVID-19 lockdown. As the unlock phases progressed, human-interventions on the GRE and its catchment increased as agriculture, aquaculture, fishing and industrial activities were allowed to resume on a phases basis. After the COVID-19 lockdown, the lack of temporal variability in nutrient status of the Muriganga was unexpected (immediately after the lockdown ended a reduced nutrient level of the estuary was expected because the human-interventions were minimum but later in the unlock phases human-interventions gradually resumed); however, interpreting the current results is hampered in the absence of any recent baseline data from regular monitoring of the nutrient status of the Muriganga.

4.2. Copepod community of Muriganga

The copepod community of the Muriganga mostly consisted of calanoids, with the Paracalanidae and Acartiidae accounting for most of the total abundance, results that are consistent with previous studies on the Muriganga section of the GRE (Sarkar et al., 1986; Paul et al., 2019). In the coastal waters of India, including the mangrove estuaries of the Indian Sundarban, the Paracalanidae plays a significant role as many species of that family build dense populations and are, therefore, crucial for the overall structure of the estuarine copepod community (Padmavati and Goswami, 1996; Fernandes and Ramaiah, 2008; Paul et al., 2019; Nandy and Modal, 2020). No significant spatial ordination of the copepod community was evident during the present study. Diversity (i.e. Shannon index and Pielou's index) did not vary between sampling sites, which is consistent with Paul et al. (2019) who observed no significant spatial change of copepod diversity in the Muriganga. Copepods such as *B. similis*, *A. tortaniformis*, *A. spinicauda*, *A. tropica*, *P. parvus*, *P. aculeatus* and *P. serricaudatus* were ubiquitously present as previously observed by Paul et al. (2019) and Paul et al. (2020b). The relative abundances of *P. indicus*, *Acrocalanus gibber*, *A. gracilis*, *A. spinicauda* and *A. sewelli* were higher in the post-lockdown period compared to their relative abundances in the pre-lockdown period. That possibly indicates, after the COVID-19 lockdown, few stenohaline species added to the community, which otherwise largely comprised euryhaline species (Paul et al., 2019; Paul et al., 2020a). Those changes, however, did not trigger any significant spatial-temporal change in the overall structure of the copepod community of the Muriganga. *Bestiolina similis* was the most abundant and dominant copepod throughout this study, a result consistent with the works of Bhattacharya et al. (2015) and Paul et al. (2020a) conducted in the mangrove estuaries of the Indian Sundarban. *Bestiolina similis* tolerates extreme and abrupt changes of salinity, water temperature and pH levels of the Muriganga and dominates the copepod community irrespective of stable or perturbed conditions in the estuary (Bhattacharya et al., 2014a; Paul et al., 2019; Paul et al., 2020a). According to the index of dominance, *B. similis*, *A. tortaniformis* and *A. spinicauda* were the dominant species in the Muriganga and their dominance index had hardly changed during the course of this study. These copepods can coexist by shifting their feeding habit according to the variable environmental conditions of an estuary (Kleppel, 1993; Djeghri et al., 2018). *Bestiolina similis* and *A. spinicauda* co-dominated the Muriganga immediately after COVID-19 lockdown ended. *Bestiolina similis* continued its dominant position in the community whereas *A. spinicauda* was replaced by *A. tortaniformis* over time from unlock 1 to 5. *Acartiella tortaniformis* is found exclusively in the coastal waters of India and tolerates a wide range of salinity, water temperature and organic loads, which may be a reason for its higher abundance in the Muriganga section of the GRE where environmental gradients fluctuate with every incoming and outgoing tide (Gajbhiye et al., 1991; Bhattacharya et al., 2014a). *Paracalanus parvus*, which is generally found in coastal, neritic and brackish waters, is a warm-water herbivorous copepod that generally feeds on diatoms (Checkley, 1980; Paul and Calliari, 2019). A study of hourly changes of the copepod

community of the Muriganga revealed that *P. parvus* has an overlapping niche with *B. similis* when the Muriganga remains ecologically stable; however, after any cyclonic disruption *P. parvus* is replaced by *A. tortaniformis* (Paul et al., 2019; Paul et al., 2020b). The cyclopoid copepods *Oithona brevicornis* and *Oithona similis* were recorded throughout the study. The relative abundance of *O. brevicornis* was significantly higher in 2020 (i.e. after the COVID-19 lockdown) than in 2019, which possibly indicate the presence of more carnivorous individuals within the copepod community, and may affect predator-prey relationships within the copepod community and the relationships of copepods with other ecological communities of the Muriganga (Bhattacharya et al., 2015; Paul et al., 2020a).

4.3. Seasonal change and the copepod community

In Indian estuaries (including that of the Sundarban) seasonal variability of the plankton community has been well studied (Padmavati and Goswami, 1996; Biswas et al., 2010; Nandy et al., 2018; Gogoi et al., 2020; Nandy and Mondal, 2020). In the Muriganga, species richness and abundance of copepods generally show two distinct peaks, in the late post-monsoon (December to middle of February) and in the late pre-monsoon (April to early June), whilst species richness generally remains relatively low in the late monsoon (August to September) (Sarkar et al., 1985, 1986). Bhattacharya et al. (2015) while studying mesozooplankton (dominated by copepods) of the Indian Sundarban supported the observation of Sarkar et al. (1986). During the monsoon, the GRE receives high nutrient inputs from its catchment (Mitra et al., 2018; Mitra et al., 2019). This helps the phytoplankton community to proliferate in the early post-monsoon, and in consequence support higher diversity and density of mesozooplankton in the late post-monsoon (Biswas et al., 2010; Bhattacharya et al., 2015; Choudhury et al., 2015; Gogoi et al., 2020). Current results contradict those of many previous studies on copepods that were conducted on or near the Muriganga and in other mangrove estuaries of the Indian Sundarban (Sarkar et al., 1985, 1986; Bhattacharya et al., 2015; Nandy et al., 2018; Nandy and Mondal, 2020). A possible explanation for the lack of seasonal peaks of species richness and abundance observed in the present study could be the effects of three successive cyclones, i.e. Fani (May 2019), Bulbul (November 2019) and Amphan (May 2020), severely disrupting the Muriganga. Successive cyclones are disruptive for the copepod community of the Muriganga, in particular for species richness and abundance of individual populations which may take a few weeks to months to recover depending on the intensity and landfall site of the cyclone (Paul et al., 2020a). It is likely that the copepod community of the Muriganga suffered, or is still suffering, from those cyclonic disruptions and that an extended monsoon of 2020 did not provide enough time for the community to stabilise and to build up the usual seasonal peaks of species richness and abundance.

4.4. Effects of abiotic variability on the copepod community

Abiotic variability affects copepod diversity and distribution in tropical river-estuaries of India (including the Sundarban region), Brazil, Malaysia and Australia (Duggan et al., 2008; Magalhães et al., 2009; Chew and Chong, 2011; Rakesh et al., 2013; Nandy and Mondal, 2020). Temporal variability of water temperature and pH, and their interactions, showed no significant impact on diversity and abundance of copepods. Such is not unusual for estuaries of the Indian Sundarban where water temperature varies only within a narrow range and the resident copepods reproduce throughout the year; therefore abundance and diversity are less affected by temperature variability of the habitat (Bhattacharya et al., 2015). If pH declines sharply that may be stressful for many copepod populations of an estuary, reducing reproductive output (egg production and brood release) of many copepods, and, therefore, likely to negatively impact diversity and abundance (Aguilera et al., 2013; Paul et al., 2020b). However, that is unlikely for copepods of the Muriganga because the habitat mostly maintains

slightly alkaline characteristics (Choudhury et al., 2015; Paul et al., 2019). The salinity of the Muriganga and species richness, abundance and Shannon index showed inverse relationships, which is not usual because those generally increase with increasing salinity of a river-estuary (Madhupratap, 1979; Attrill, 2002; Lawrence et al., 2004; Paul et al., 2016). The extended monsoon of 2019–2020 and three successive cyclones during this study interval possibly kept the overall salinity profile of the estuary within the mesohaline range, which otherwise remains within the polyhaline range except for the months of the monsoon when a mesohaline condition is common in the Muriganga (Mukhopadhyay et al., 2006; Choudhury et al., 2015; Paul et al., 2019; Paul et al., 2020a). Under such circumstances, many estuarine-specialist copepods such as *B. similis*, *A. tortaniformis*, *A. spinicauda*, *A. tropica*, *P. parvus*, *P. aculeatus* and *P. serricaudatus* thrive and may build large populations. So, the total abundance remained high despite a mesohaline profile of the Muriganga. Copepods such as *P. dubia*, *A. tropica*, *A. sewelli*, *P. serricaudatus* and *A. tortaniformis*, which generally reside within the oligohaline zone of the Muriganga, have possibly invaded the mesohaline zone, increasing the diversity despite an overall low salinity profile.

4.5. Copepod community as indicator of ecological stress of estuaries

Estuarine ecological studies have traditionally focused on the patchy distribution of copepods, presence/absence of certain species, values of various diversity indices, and their relationships with environmental gradients, to draw inferences on the ecological stress of estuaries (Lawrence et al., 2004; Rice et al., 2015; Paul et al., 2016; Paul and Calliari, 2019). Changes of diversity, abundance and distribution of copepods are, therefore, considered as indicators of ecological stresses of estuaries that arise from abiotic, seasonal, climate changes and human-interventions (Sinha et al., 1996; Hoof and Peterson, 2006; Sullivan et al., 2007; Lin et al., 2011; Paul et al., 2016; Araujo et al., 2017). Seasonal and inter-annual variability of copepods has been studied in detail for predicting the impacts of climate change on estuarine food webs and its implications for commercial fisheries that exploit copepods as food (Beaugrand and Ibanez, 2004; Hoof and Peterson, 2006; Sullivan et al., 2007; Rice et al., 2015). Previous studies on tidal, daily, monthly, seasonal, inter-seasonal and inter-annual variability of diversity, abundance and dominance of mesozooplankton (including copepods) of the GRE and the Indian Sundarban have indicated the ongoing decadal change (a switch from a herbivore-dominated to an omnivore dominated community) and climate change (e.g. warming) of estuaries in the region (Sarkar et al., 1986; Sinha et al., 1996; Bhattacharya et al., 2015; Nandi et al., 2018; Paul et al., 2019; Nandi and Mondal, 2020). The COVID-19 lockdown was a rare opportunity when the impacts of many human activities that are detrimental to most forms of aquatic life temporarily halted (or slowed down in some cases); therefore, an expectation of higher diversity and abundance within the plankton community is not unrealistic. Results indicated, however, that abiotic variability, seasonal change, and the COVID-19 lockdown did not greatly affect the copepod community of the Muriganga. Few species changed their presence/absence and relative abundance over the course of this study. The community; however, maintained its structure without being much disturbed by any habitat variability. It is thus important to assess the flexibility of an ecological community before concluding on its vulnerability to abiotic variability, seasonal fluctuations and any change in the intensity of human-intervention (Bhattacharya et al., 1994; Sinha et al., 1996; Bhattacharya et al., 2014b; Paul et al., 2020a). The copepods *B. similis*, *A. spinicauda*, *A. tortaniformis* and *P. parvus* are among those common estuarine-resident species that have demonstrated flexibility to adapt to extreme and abrupt changes of the GRE in the face of cyclonic disruptions and within days have recolonised their respective micro-habitats and built

large populations (Bhattacharya et al., 2014a; Paul et al., 2020b). The COVID-19 lockdown was a period of minimum human-intervention so it was presumed that diversity indices of the copepod community would peak immediately after the COVID-19 lockdown ended and then diversity indices would gradually decline because human-interventions gradually resumed in the unlock phases. Contrary to that, after the COVID-19 lockdown, species richness and Shannon index of the copepod community were higher in the unlock 5 phase compared to their respective levels in the unlock 1 phase. A possible explanation of that result is that the extremely severe cyclonic storm Amphan (1 minute sustained windspeed 260 km/hour) made landfall near the mouth of the GRE and ravaged the Muriganga and its ecological communities (experienced windspeed of 150–160 km/hour) on 20 May 2020 (i.e. 11 days before the end of the COVID-19 lockdown). The cyclone Amphan possibly disrupted many ecological communities of the GRE for months, so that species richness and total abundance of the copepod community remained low in the unlock 1 to 2 phases. The higher species richness and diversity observed from September 2020 onwards (i.e. in the unlock 4 to 5) are possibly more related to post-cyclone recovery of the copepod community than to changes of habitat that are directly or indirectly related to the COVID-19 lockdown. The relative abundance of a few copepod species did increase in the post-lockdown period compared to their pre-lockdown status, but those changes have not brought any significant structural changes within the copepod community. Overall, the copepod community remained mostly indifferent to many forms of habitat variability of the GRE that occurred between February 2019 and October 2020. That is plausible when an ecological community has high flexibility to overcome habitat variability that is natural and/or man-made. Therefore, by studying the structural change of that community for a few seasons to a few years, many ecological stresses of excessively large river-estuaries may go unnoticed.

Declarations

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Tables

Sampling Interval	Salinity			Water temperature (°C)			pH			
	S1	S2	S3	S1	S2	S3	S1	S2	S3	
Pre-lockdown	Median	12.25	12.60	12.59	23.40	23.30	23.30	8.10	8.11	8.14
	Range	8.90-19.30	8.85-19.20	8.60-19.20	20.40-31.20	20.70-31.10	20.70-31.30	6.10-8.64	6.10-8.60	6.20-8.60
Post-lockdown	Median	16.20	16.30	16.30	29.70	29.90	29.90	8.31	8.22	8.25
	Range	5.10-18.30	5.10-18.20	4.80-19.30	28.30-30.20	28.30-30.80	28.40-30.20	6.00-8.41	7.66-8.56	7.76-8.51

Table 1. Site-specific variations of salinity, temperature and pH levels of Muriganga section of Ganges River estuary before (February-December 2019) and after (June-October 2020) COVID-19 lockdown (24 March to 31 May 2020) of India.

Table 2. Relative abundance (%) of copepod species sampled from three sampling sites (S1, S2, and S3) on the Muriganga section of the Ganges River estuary before (February to December 2019) after (June to October 2020) the COVID-19 lockdown (24 March to 31 May 2020) of India.

Copepod species	S1		S2		S3	
	Pre-lockdown	Post-lockdown	Pre-lockdown	Post-lockdown	Pre-lockdown	Post-lockdown
<i>Paracalanus parvus</i>	12.86	09.62	10.60	08.30	09.53	08.69
<i>P. aculeatus</i>	04.77	03.01	04.79	03.19	04.14	04.09
<i>P. indicus</i>	00.47	03.79	01.18	04.49	01.02	04.09
<i>Parvocalanus dubia</i>	02.03	00.78	02.13	03.70	04.40	03.49
<i>Acrocalanus gibber</i>	00.00	02.26	00.00	00.59	00.00	00.00
<i>A. gracilis</i>	01.69	04.24	02.75	03.77	0.86	02.90
<i>A. longicornis</i>	01.86	01.50	00.74	02.52	00.90	02.36
<i>Bestiolina similis</i>	21.30	20.47	21.73	19.00	19.45	16.60
<i>Parvocalanus crassirostris</i>	03.26	04.42	02.47	01.33	01.53	03.71
<i>Acartia spinicauda</i>	08.70	13.84	09.83	13.34	10.31	12.59
<i>Acartiella sewelli</i>	00.00	02.37	00.00	02.65	00.00	02.32
<i>Acartia tonsa</i>	02.33	00.00	01.40	00.61	01.17	01.80
<i>A. tropica</i>	04.59	00.00	04.56	02.98	06.63	03.41
<i>Acartiella tortaniformis</i>	16.65	17.50	11.31	14.47	13.13	11.59
<i>Pseudodiaptomus binghami</i>	02.50	01.41	02.49	00.59	02.90	00.56
<i>P. serricaudatus</i>	02.96	03.64	06.84	01.22	08.87	04.65
<i>Canthocalanus pauper</i>	00.00	01.41	02.10	02.39	02.55	01.66
<i>Eucalanus subcrassus</i>	00.47	00.82	03.03	01.80	01.15	01.69
<i>E. crassus</i>	01.55	00.00	01.41	00.00	00.86	00.58
<i>Labidocera euchaeta</i>	01.08	00.00	01.40	00.00	00.86	00.00
<i>L. acuta</i>	01.57	00.72	01.76	01.80	01.17	00.58
<i>Temora turbinata</i>	00.00	01.41	01.24	00.61	00.76	01.80
<i>Corycaeus danae</i>	01.08	00.82	01.03	03.03	00.86	01.10
<i>Oncaea venusta</i>	03.16	00.00	00.00	00.00	00.00	02.36
<i>Oithona similis</i>	00.47	00.00	00.36	00.61	00.61	00.00
<i>O. brevicornis</i>	04.68	05.98	04.85	07.01	06.35	07.39

Table 3. Total abundance (ind.m⁻³) and species richness of copepods sampled before (February 2019 to December 2019) and after (June to October 2020) the COVID-19 lockdown (24 March to 31 May 2020) of India from three sampling sites (S1, S2 and S3) on the Muriganga section of the Ganges River estuary.

Sampling Period	Sampling Date	S1		S2		S3	
		Total Abundance	Species Richness	Total Abundance	Species Richness	Total Abundance	Species Richness
	25/02/2019	36100	19	35610	14	35970	13
	14/05/2019	46000	07	67000	13	83000	15
Pre-lockdown	24/08/2019	29166	17	30000	19	30833	16
	18/11/2019	51333	09	74733	12	88000	11
	28/12/2019	13333	08	21667	09	23767	09
	05/06/2020	28000	08	54833	10	55533	11
Post-lockdown	12/06/2020	44433	12	59950	12	68200	16
	09/09/2020	39600	13	63783	17	86667	20
	03/10/2020	72000	17	73600	20	100800	22

Sampling Period	Sampling Date	<i>Bestiolina similis</i>	<i>Acartiella tortaniformis</i>	<i>Acartia spinicauda</i>	<i>Paracalanus parvas</i>	<i>Oithona brevicornis</i>
	25/02/2019	0.12	0.01	0.06	0.08	0.06
	14/05/2019	0.23	0.18	0.13	0.11	0.04
Pre-lockdown	24/08/2019	0.13	0.06	0.04	0.11	0.02
	18/11/2019	0.22	0.15	0.09	0.11	0.08
	28/12/2019	0.29	0.28	0.11	0.07	0.02
	05/06/2020	0.22	0.14	0.21	0.07	0.08
Post-lockdown	12/06/2020	0.19	0.14	0.12	0.11	0.08
	09/09/2020	0.17	0.14	0.10	0.08	0.07
	03/10/2020	0.16	0.13	0.11	0.07	0.05

Table 4. Index of dominance of few frequently sampled copepod species of the Muriganga section of the Ganges River estuary before (February to December 2019) and after (June to October 2020) COVID-19 lockdown (24 March to 31 May 2020) of India.

Figures

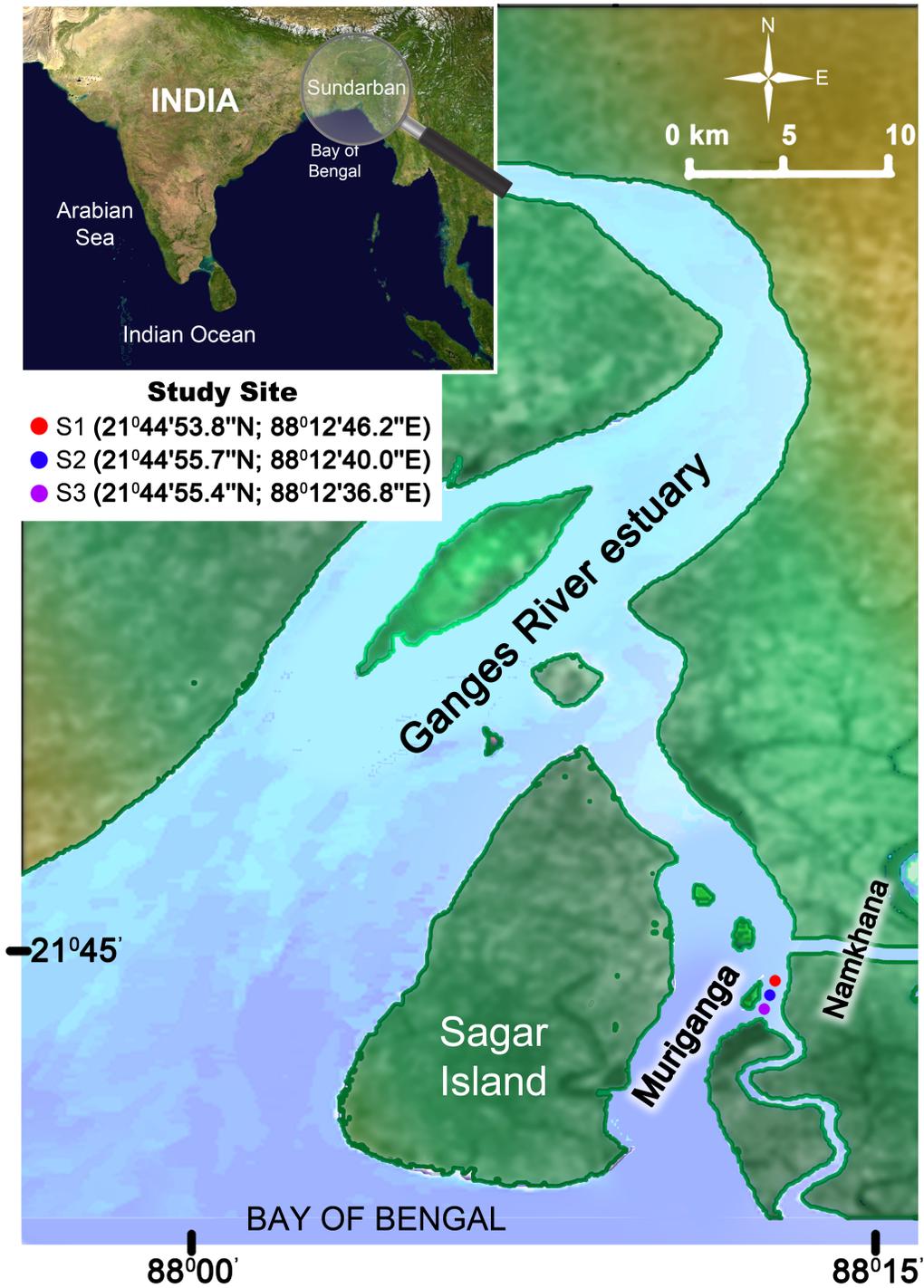


Figure 1

Study sites (i.e. S1, S2 and S3) on the Muriganga section of Ganges River estuary of India.

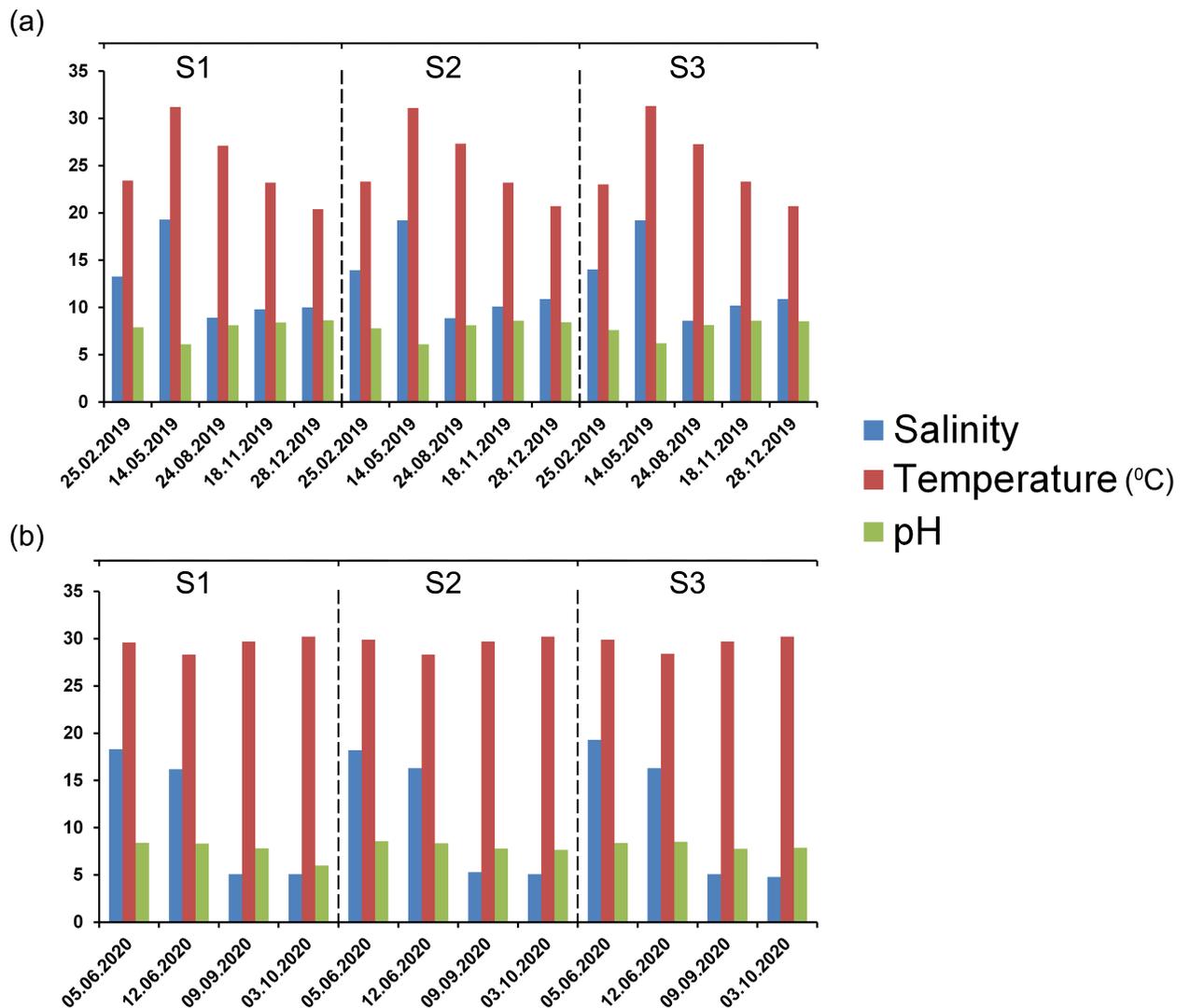


Figure 2

Site-specific variation of salinity, water temperature (°C) and pH levels of the Muriganga section of the Ganges River estuary. a = Pre-lockdown (February 2019 to December 2019), b = Post-lockdown (June 2020 to October 2020).

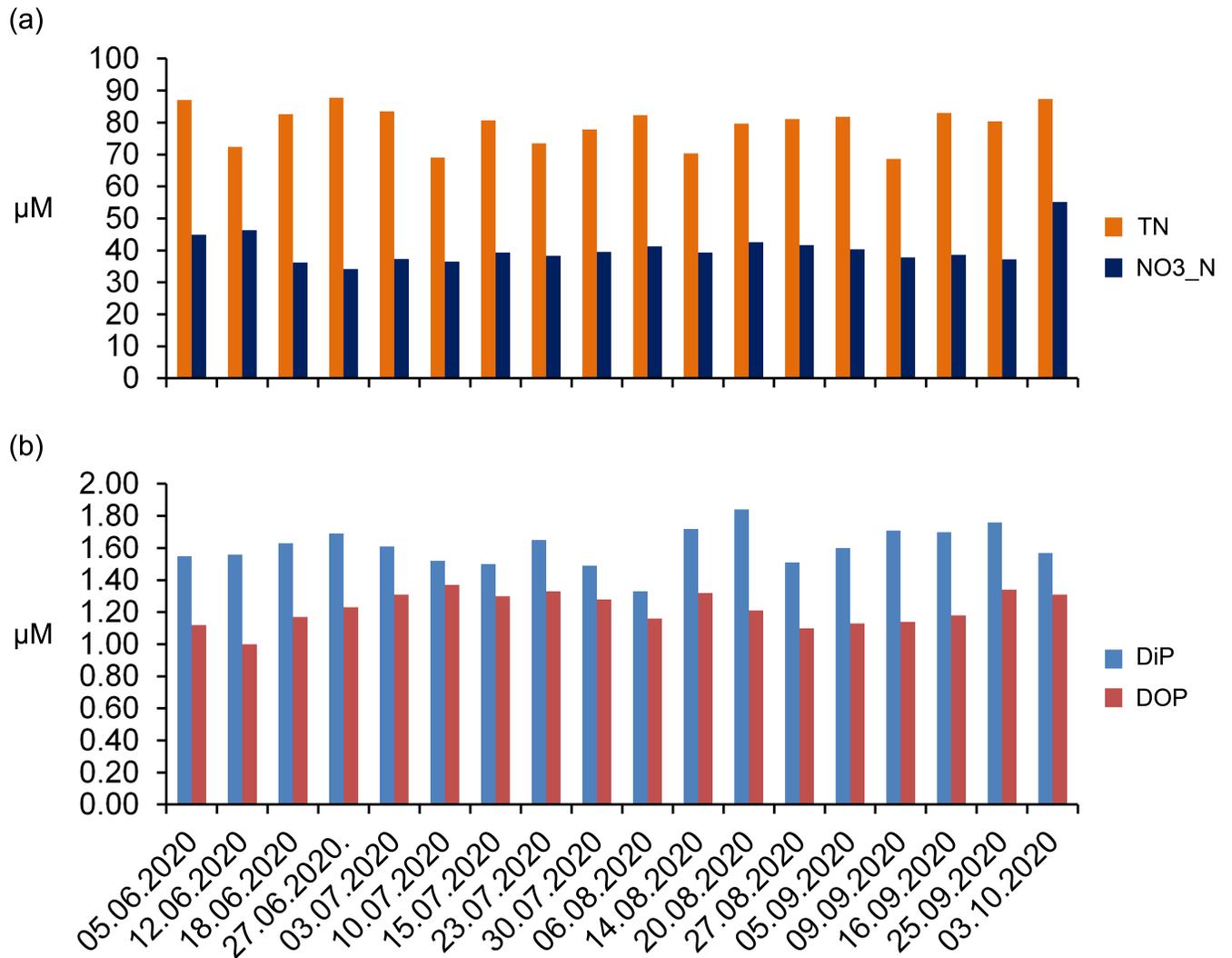


Figure 3

Nutrient concentrations (μM) of the Muriganga section of the Ganges River estuary between unlock phases 1 to 5 (June 2020 to October 2020). a = Total Nitrogen (TN) and Nitrate-Nitrogen ($\text{NO}_3\text{-N}$), b = Dissolved Inorganic Phosphate (DiP) and Dissolved Organic Phosphate (DOP)

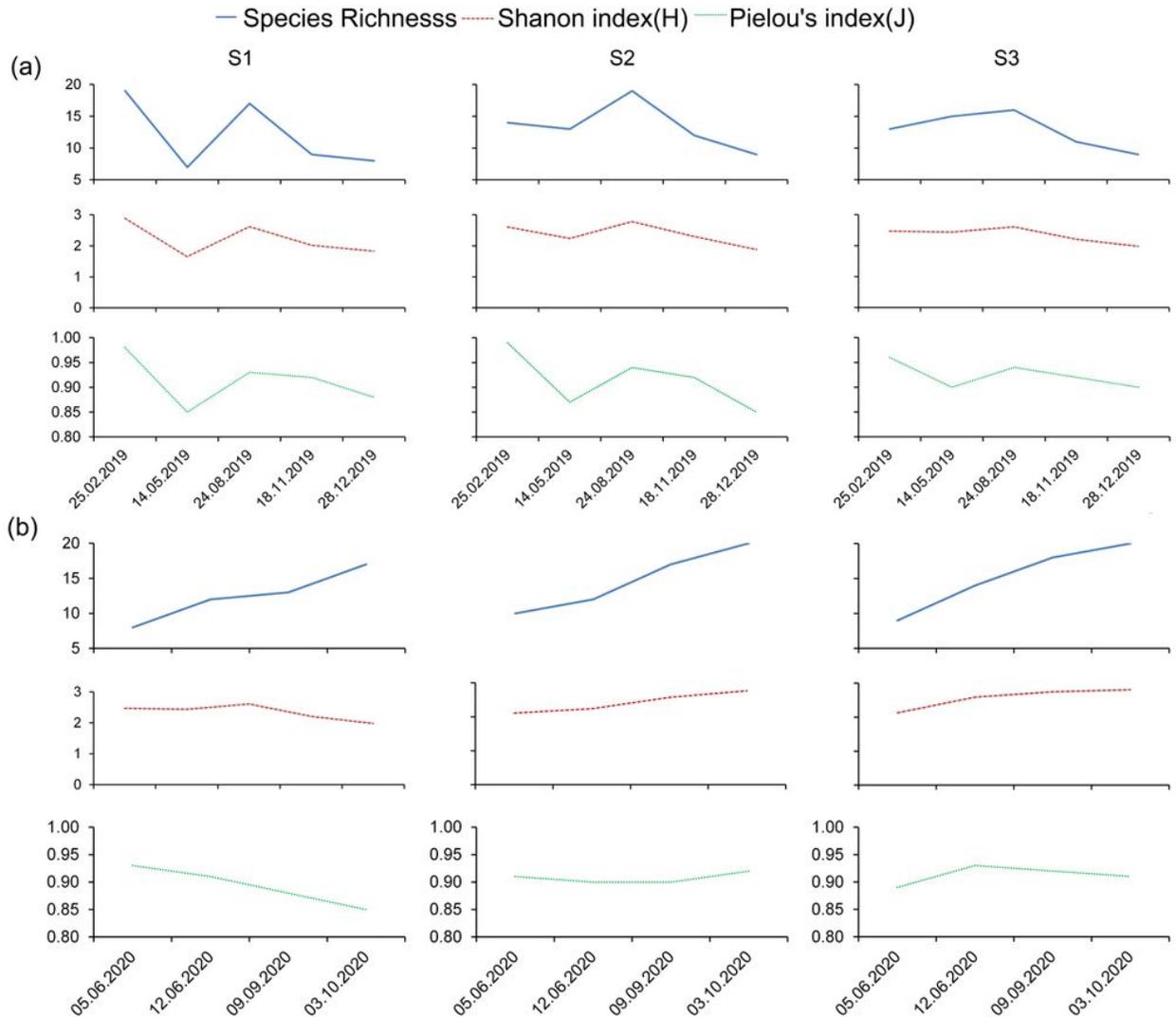


Figure 4

Spatial-temporal variations of species richness, Shannon index (H) and Pielou's index (J) of the copepod community sampled from the Muriganga section of the Ganges River estuary. a = Pre-lockdown (February 2019 to December 2019), b = Post-lockdown (June 2020 to October 2020).

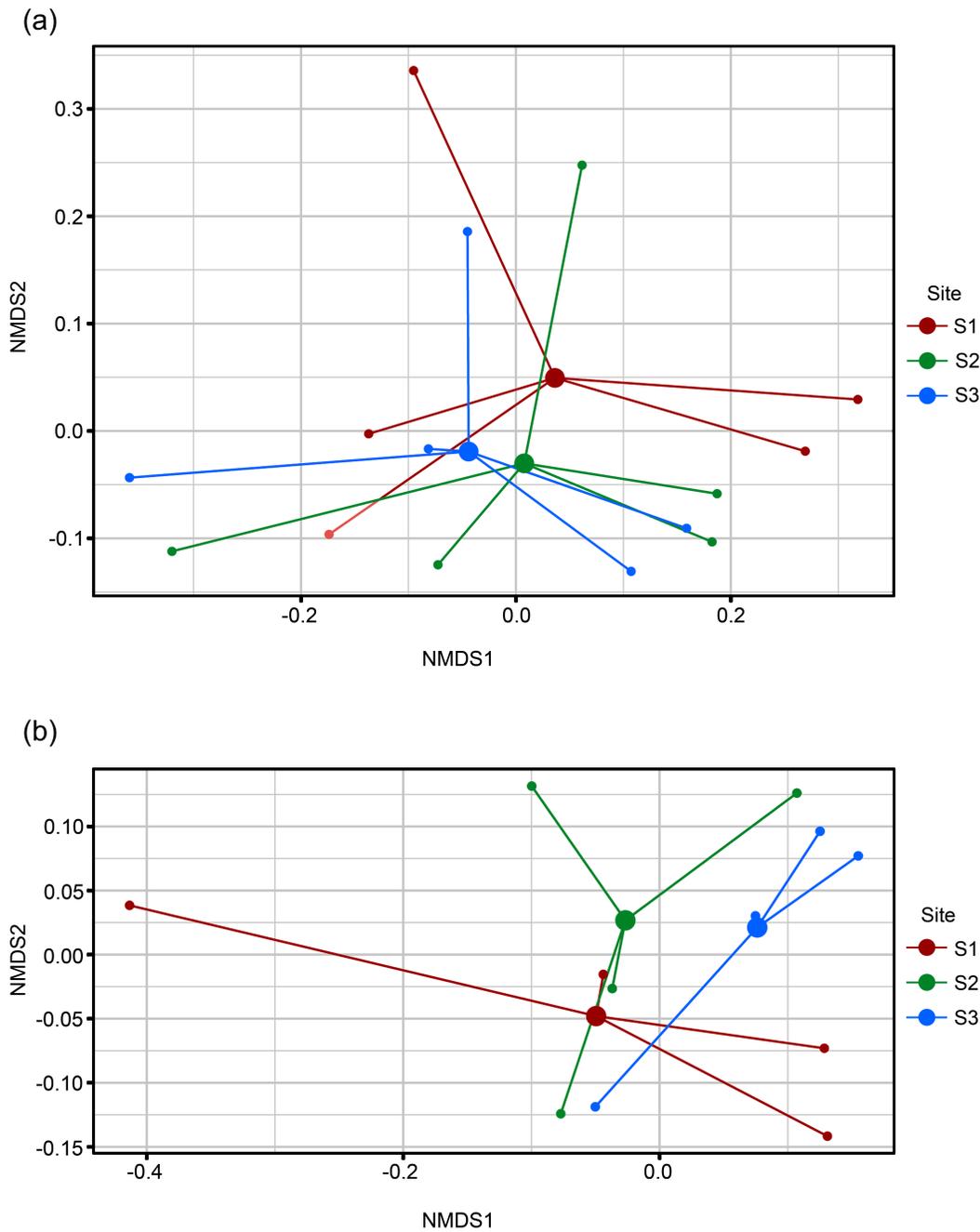


Figure 5

Non-metric Multidimensional Scaling (NMDS) plots using the Bray–Curtis measure of dissimilarity exhibiting spatial ordination of the copepod community of the Muriganga section of the Ganges River estuary. a = Pre-lockdown period (February 2019 to December 2020) and b = Post-lockdown period (June 2020 to October 2020).

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