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Distribution and community structure of microphytoplankton in relation to increasing anthropogenic impact along coastal waters of Jeddah, the central Red Sea

by

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Abstract

Seasonal distribution and diversity of surface phytoplankton have been studied in relation to anthropogenic activities at seven locations along the coastal waters of Jeddah, Saudi Arabia in 2017. The concentration of nitrite, nitrate, ammonia, phosphate and silicate varied over wide ranges: 0.04–20.27, 0.05–29.3, 0.22–78.16, 0.02–25.90 and 43–24.50 μmol l^{-1}, respectively. Inorganic nutrients, phytoplankton biomass and density were at eutrophic levels at two lagoon locations (4 and 5), while other coastal locations showed oligotrophic characteristics of the Red Sea. Phytoplankton biomass was positively correlated with all measured inorganic nutrients. The total phytoplankton density varied between 52.4×10^3 and 40800×10^3 cells m^{-3} (average $6249.9 \times 10^3 \pm 10\,797 \times 10^3$ cells m^{-3}). A total of 174 species of phytoplankton (95 diatoms, 75 dinoflagellates, 3 cyanophytes and 1 silicoflagellates) were recorded in this study, with the dominance of diatoms (95.1%). Higher densities observed at lagoon locations were mainly due to the proliferation of the diatom species *Skeletonema costatum* and *Chaetoceros decipiens*. In terms of diversity, dinoflagellates were more numerous than diatoms in July, otherwise mostly diatoms dominated. On the other hand, cyanophytes were more abundant in November. Based on the present study, anthropogenic activities (especially sewage effluent) in Jeddah coastal waters had a significant impact on the phytoplankton densities and diversity.

Key words: nutrients, phytoplankton, coastal water, Jeddah, Red Sea

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Introduction

Phytoplankton are known as microalgae that are well adapted to life in several aquatic ecosystems such as rivers, ponds and seas (Reynolds 2006). They play an important role in such ecosystems by forming the base of the food web and perform a major portion of the primary production, thereby maintaining the nutrient recycling within the system (Dawes 1998; Mann 1999; Reynolds 2006). They also play an important role in the sequestration of carbon by capturing CO_2 though the process of photosynthesis and sinking it toward the deeper parts of the ocean after their death, thus reducing its concentration in the atmosphere and, in turn, helping to fight global warming (Falkowski et al. 2004). Many factors, such as light, macro- and micronutrients, determine the successful proliferation of phytoplankton in the ecosystems (Banse 1992). In aquatic ecosystems, the effects of spatio-temporal heterogeneity are particularly strong, as physical and biological processes are interlinked with each other both in spatial and seasonal aspects (Steele 1985). Over the past 40 years, the human activity has increased the flux of nitrogen and phosphorus, mainly from estuaries, sewage treatment plants and agricultural activities, to the coastal waters about two and three times, respectively (Howarth et al. 2002).

Marine eutrophication is now considered as an ecological problem, which is increasing in its magnitude and creating potential impacts on the coastal ecosystems worldwide (McIntyre 1995; Nixon 1995; Pearl et al. 1997). The main cause of this phenomenon is the introduction of a higher concentration of inorganic nutrients into the system through various anthropogenic sources. This can severely affect the ecosystem, by breaking the ecological balance, as well as the seasonality of ecosystem functions. Several man-made problems, such as the release of heavy metals, pesticides, persistent organic pollutants into the system as well as the most dangerous oil spills, have a negative impact on the growth and multiplication of phytoplankton in both coastal and open ocean regions (Echeveste et al. 2011; Huang et al. 2011; Matthews 2013). Many of these processes affect the phytoplankton productivity. The increasing environmental pollution is another aspect that affects the phytoplankton dynamics mainly in coastal marine ecosystems (Häder & Gao 2015). The primary producers are easily the major target of environmental pollution, which in turn affects the higher trophic organisms. Sudden changes in water quality, such as high nutrient loading or temperature, are responsible for shifting the aquatic food webs more toward bacteria, viruses and nano/picoplankton activities. These situations are more likely to cause harmful effects than positive ones (Pinckney et al. 2001).

The Red Sea is generally considered an oligotrophic ecosystem. Compared to other marine ecosystems, it has received little attention in studies dealing with phytoplankton community composition. In the Red Sea, the Saudi Arabian coastal waters were even less explored in this respect (Dowidar 1983; Khalil et al. 1984; Shaikh et al. 1986; Khalil 1988; Touliabah et al. 2010). There are a number of studies that are focused on the composition of the phytoplankton community in the western coastal waters of the Red Sea and the Gulf of Aqaba, which is the northward extension of the Red Sea proper (e.g. Post et al. 1996; 2002; El-Sherif & Aboul Ezz 2000; Al-Najjar et al. 2007; Madkour et al. 2010; Nassar & Khairy 2014; Abbass et al. 2018; El-Sherbiny et al. 2019). Despite the oligotrophic characteristics of the Red Sea, the incorporation of various anthropogenic inputs in few coastal locations results in the formation of eutrophic patches among oligotrophic environments. Therefore, the study was carried out primarily to determine the spatio-temporal variation of phytoplankton and most importantly to assess the role of anthropogenic inputs on the phytoplankton community along the coastal waters of Jeddah, the central Red Sea, Saudi Arabia.

Materials and methods

Study area

The present study was carried out in the coastal waters of Jeddah, which is a fast-growing metropolitan city located by the central Red Sea. This region provides many opportunities to study the effect of anthropogenic impact on the coastal marine ecosystems of the Red Sea. The presence of different human interferences makes this region vulnerable to environmental changes that can eventually affect biotic and abiotic interactions within the system. In order to study such changes, seven sites were selected along the Jeddah coastline (Fig. 1). They were selected in such a way that they ultimately represent various regions in the system, which are not similar in terms of anthropogenic interference. Site 1 was located in the northern part of Jeddah, where the magnitude of human interference is minimal and can be considered as a reference location for the study. Site 2, located in the central region, is exposed to human impact through wastewater, while site 3, also located in the central region, is mainly affected by the presence of a desalination plant. Sites 4 and 5 represent in the

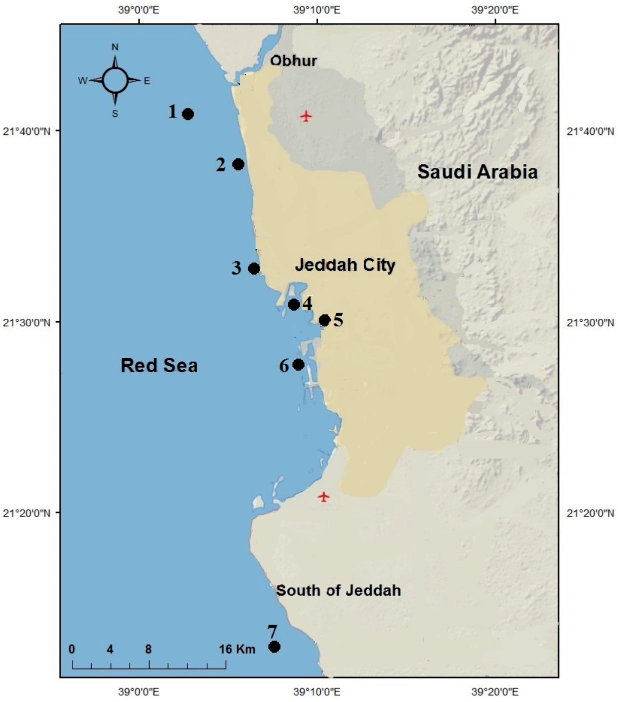

Figure 1

Map showing the sampling locations in neritic water around Jeddah, Saudi Arabia

current study the most polluted areas, which are called Al-Shabab and Al-Arbaeen lagoons, respectively. Both sites are exposed to human intervention in many ways, mainly because of their location in the heart of the city. Site 6 was located in the active zone of the Jeddah international port, which can be considered as one of the largest ports functioning on the entire Red Sea coast. The last site (7) was located further south, near a sewage treatment plant.

Field sampling and analysis

The fieldwork was carried out in March, July and November 2017 using a mechanized boat. Physical parameters (such as surface temperature and salinity) were determined using a multiparameter water quality probe (Horiba U50). In order to determine inorganic nutrients and phytoplankton biomass (chlorophyll *a*), a water sample was collected from a depth of 5 m depth using a Niskin sampler (5 l). At least 10 l of seawater was collected and kept in a cool, dark place until reaching the laboratory. After reaching the shore laboratory, 8–10 l of seawater was filtered through a filter paper with a pore size of 0.7 µm and a diameter of 47 mm (Whatman GF/F) to determine the content of chlorophyll *a*. After the filtration, a few drops of magnesium carbonate ($MgCO_3$) were added to the filter paper to prevent the disintegration of chlorophyll molecules. All filter papers were then wrapped in aluminum foil and kept in a deep freezer (–80°C). To determine inorganic nutrients, about 1 l of seawater was filtered through a membrane filter paper with a pore size of 0.45 µm, and the filtered seawater was then placed in a deep freezer until further analysis. Both chlorophyll *a* and inorganic nutrients were determined according to the protocols specified by Parsons et al. (1984) using a UV spectrophotometer (Shimadzu UV 1700). Phytoplankton samples were collected using a plankton net with a mesh size of 20 µm (Hydrobios) equipped with a flowmeter to determine the volume of water filtered. Horizontal towing (0.5 m below surface water) was carried out for 5 min at a speed of about 1 knot. Immediately after the collection, the phytoplankton samples were preserved with Lugol's iodine solution and a few drops of concentrated formaldehyde solution in an amber glass bottle (Kürten et al. 2015). All samples were then stored in a cool, dark place until further analysis. Triplicate phytoplankton counts were carried out in a Sedgewick Rafter counting chamber using an inverted microscope (Leica DMI 3000B) according to the standard procedures of LeGresley & McDermott (2010). Phytoplankton species were taxonomically identified with the help of identification catalogues (Taylor 1976; Tomas 1997; Gómez 2013). They were then validated with the help of WoRMS (World Register of Marine Species, www.marinespecies.org) and named according to the latest taxonomic nomenclature.

Data interpretation and statistical analysis

Interpretation of the obtained data was carried out using statistical software SPSS V23, and PRIMER 6. In order to determine the relationship between different physical, chemical and biological parameters, Pearson's coefficient of correlation (*r*) was calculated. Moreover, one-way analysis of variance (ANOVA) was carried out to determine both spatial and temporal variation of different parameters. Biodiversity indices (Margalef's species richness, Pielou's evenness index and Shannon diversity index) were calculated to study the differences in the phytoplankton spatial and temporal distribution. Nonmetric multidimensional scaling (NMDS) was carried out to determine the similarity between different sites based on various environmental parameters. The dendrogram based on the Bray-Curtis similarity index of the phytoplankton abundance was constructed to determine similarities between different sampling locations.

Results

Physicochemical parameters

Temperature ranged between 24.5°C at site 1 and 30.1°C at site 4 in March and July, respectively, with minimal spatial variation. High average temperature (29.5°C) was recorded in July at all sites. The surface salinity showed significant differences between the sites ($p < 0.05$), especially the lagoon sites (4 and 5), which showed lower values (on average 35.2 and 36.1, respectively) during the study period (Fig. 2). The maximum salinity of 39.9 was observed at site 3 in July, while the minimum (35.1) was observed at site 4 in July (Fig. 2). During the present study, the concentrations of all nutrient salts showed significant spatial variability ($p < 0.05$) with higher values at sites 4 and 5 compared to the other sites (Fig. 3). The nitrite concentration fluctuated between 0.04 and 20.27 μmol l^{-1} at sites 1 and 5 in July, respectively, with an average value of 3.93 ± 6.82 μmol l^{-1} (Fig. 3a). These values were relatively high in November (average 4.84 ± 8.2 μmol l^{-1}). Nitrate concentrations ranged between a minimum of 0.05 μmol l^{-1} at site 2 and a maximum of 29.30 μmol l^{-1} at site 4 in July with an average value of 4.33 ± 6.82 μmol l^{-1} (Fig. 3b). The highest average value of 6.20 ± 11.37 μmol l^{-1} was recorded in November, while the lowest in July. Ammonium was the largest source of inorganic nutrient derivatives for the studied months, mainly at the lagoon sites, but to a minimum extent also at the coastal sites. Its average concentration was 15.47 ± 26.63 μmol l^{-1}. The minimum concentration was observed at site 7 (0.22 μmol l^{-1}) and the maximum of 78.16 μmol l^{-1} at site 5 in July (Fig. 3c). Its average seasonal values ranged between 13.15 and 18.24 μmol l^{-1} in March and November, respectively. Higher average concentrations of phosphate were observed in November (5.35 μmol l^{-1}), with a maximum of 25.90 μmol l^{-1} at site 4 (Fig. 3d). On the other hand, its minimum value (0.02 μmol l^{-1}) was observed in July at site 3 (average 2.62 μmol l^{-1}). Silicate concentrations did not show any seasonal variation and ranged between 0.43 μmol l^{-1} at site 6 and 24.50 μmol l^{-1} at site 4 in November with an overall average of 6.49 ± 8.38 μmol l^{-1} (Fig. 3e).

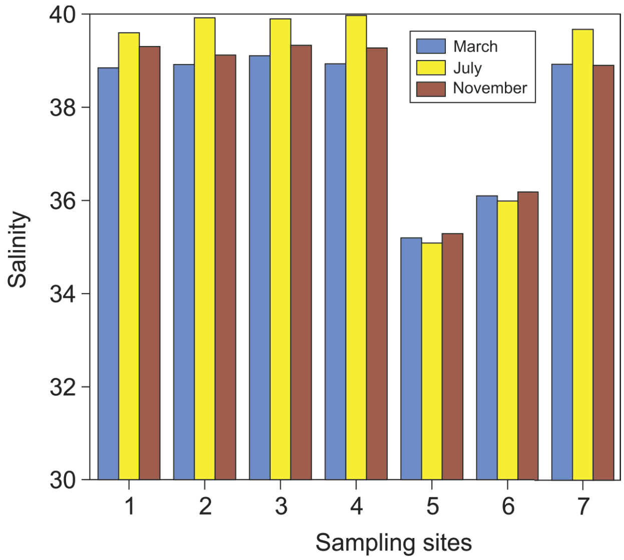

Figure 2

Changes in surface water salinity (PSU) at the sampling sites during the study period

The phytoplankton biomass in terms of chlorophyll *a* showed a significant variation among the sites ($p < 0.05$). At the coastal sites (1–3 and 6–7), the biomass followed mainly the oligotrophic nature of the Red Sea, while at the lagoon sites (4 and 5) it showed an opposite trend, indicating almost eutrophic conditions (Fig. 3f). The lowest phytoplankton biomass was determined at site 1 (0.13 mg m^{-3}) in July and the highest at site 4 (269.14 mg m^{-3}) in November with an average value of 41.03 ± 51.31 mg m^{-3} (Fig. 3f). Irrespective of the site, the highest average value of 56.59 mg m^{-3} was recorded in November, while the lowest concentration of 29.73 mg m^{-3} in March.

Population density, distribution and species composition

The total phytoplankton abundance varied significantly between the sites ($p < 0.05$), ranging from a minimum of 52.4 × 10^3 cells m^{-3} at site 1 in July to 40 800 × 10^3 cells m^{-3} at site 4 in November, with an overall average of 6249.9 × 10^3 ± 10 797 × 10^3 cells m^{-3}. The lagoon sites (4 and 5) showed the highest phytoplankton densities in all months of sampling with mean values of 23 638.2 × 10^3 cells m^{-3} and 16 469 × 10^3 cells m^{-3} at sites 4 and 5, respectively (Fig. 4a). Other sites showed considerably lower numbers, ranging from 52.4 × 10^3 cells m^{-3} at site 1 in July to 3507.7 × 10^3 cells m^{-3} at site 2 in November (Fig. 4a). In the case of the lagoon sites, the phytoplankton density fluctuated between 6544.7 × 10^3 and 40 800 × 10^3 cells m^{-3} at site 4 in July and November, respectively (Fig. 4a). On the other hand, less pronounced fluctuations in the total density of phytoplankton were found at other coastal sites (1–3 and 6–7). As for the seasonal variation in the phytoplankton density, November showed the highest value (on average 9937.2 ± 15 144.5 × 10^3 cells m^{-3}), while the lowest average value was recorded in July.

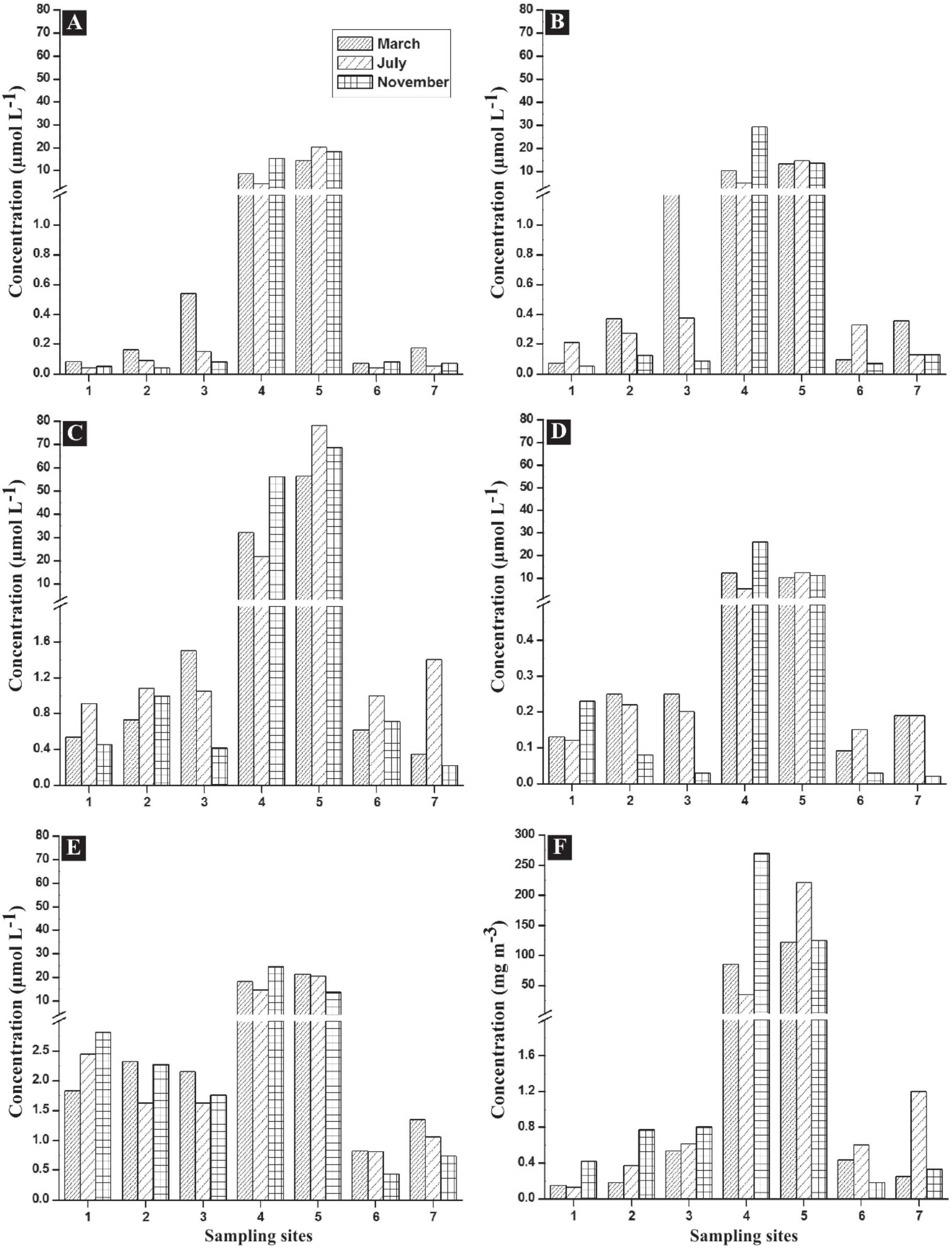

Figure 3

Nutrient salts (µmol l^{-1}) and chlorophyll *a* (mg m^{-3}) concentrations at the sampling sites during the study period: A) nitrate, B) nitrite, C) ammonia, D) phosphate, E) silicate and F) chlorophyll *a*

A total of 174 species of phytoplankton were identified in this study (95 diatoms, 75 dinoflagellates, 3 cyanophytes and 1 silicoflagellates; Table 1). Diatoms were the most dominant taxa, accounting for 95.1% of the total phytoplankton at all sites and varied from 15.5 to 40 600 × 10^3 cells m^{-3} at sites 7 and 5 in November, respectively (average 5943.2 × 10^3 ± 10 717.2 × 10^3 cells m^{-3}; Fig. 4b). They were followed by dinoflagellates, which constituted 3.3% of the total phytoplankton density and varied between 6.6 × 10^3 cells m^{-3} at site 1 in March and 2066.7 × 10^3 cells m^{-3} at site 5 in July with an average value of 247.5 ± 276×10^3 cells m^{-3} (Fig. 4c). On the other hand, the density of cyanophytes was minimal (average: 99.2 × 10^3 ± 224.1 × 10^3 cells m^{-3}), with relatively high abundance at site 6 in November (Fig. 4d).

Five diatom species accounted for about 89.2% of the total phytoplankton population, namely: *Skeletonema costatum* (Greville) Cleve, 1873 (48.5%), *Chaetoceros decipiens* Cleve, 1873 (32.4%), *Coscinodiscus radiates* Ehrenberg, 1840 (3.7%), *Chaetoceros curvisetus* Cleve, 1889 (2.8%) and *Chaetoceros affinis* (1.8%). These species occurred with very high abundance at the lagoon sites (4 and 5). In March, the centric diatom species *C. decipiens* was the most dominant species (average: 121.7 × 10^3 cells m^{-3}), accounting for about 24.5% of the total phytoplankton. This high value resulted in a higher density (8500 × 10^3 cells m^{-3}) at site 5, which contributed 79.5% to the total phytoplankton density at this site.

Furthermore, *C. curvisetus* contributed about 19% to the total phytoplankton density at site 5 (2100 × 10^3 cells m^{-3}). On the other hand, *S. costatum* occurred with the maximum density of 23 500 × 10^3 cells m^{-3} (99.7% of the total phytoplankton) at site 4. In July, *C. decipiens* and *C. radiates* were the main diatom species responsible for the higher densities at sites 4 and 5, respectively. In addition, *C. affinis* and *C. curvisetus* occurred at site 5 in relatively large numbers – 1666.7 × 10^3 and 1000 × 10^3 cells m^{-3}, accounting for 8.7% and 5.2% of the total density, respectively. It is worth mentioning that the harmful dinoflagellate *Alexandrium tamiyavanichii* Balech, 1994 dominated (1200 × 10^3 cells m^{-3}, accounting for 6.2% of the total phytoplankton) compared to the other dinoflagellate species. The massive bloom of *S. costatum* (40 000 × 10^3 cells) and *C. decipiens* (18 899 × 10^3 cells m^{-3}) was observed at sites 4 and 5, respectively, in November. The blue green alga *Trichodesmium erythraeum* dominated at coastal sites 1–3 and 6–7 (average: 399 × 10^3 cells m^{-3}) in November and contributed 2.9% to the total population.

The coastal sites showed different patterns of the diatom species dominance: *T. erythraeum*, *Chaetoceros* sp., *Thalassiothrix longissimi* Cleve & Grunow, 1880, *Pseudo-nitzschia delicatissima* (Cleve) Heiden, 1928, *C. decipiens* and *Guinardia flaccida* (Castracane) H.Peragallo, 1892 accounted for 18.62%, 9.24%, 6.16%, 4.09%, 3.55% and 2.30% of the total phytoplankton, respectively. *Tripos furca* (Ehrenberg) F.Gómez, 2013 and *Protoperidinium oceanicum* (VanHöffen, 1897) Balech, 1974 dominated among the dinoflagellates and they represented 1.52 and 1.45% of the total phytoplankton at the coastal sites. The diatom flora comprised 67 Centrales and 28 Pennales. The centric diatom genus *Chaetoceros* dominated the diatom community with a representation of 20 different species. It was followed by another centric genus, *Rhizosolenia*, represented by 15 species, and then eight species of the pennate diatom *Pseudo-nitzschia*. Whereas, *Thallassiothrix* and *Guinardia* were represented by four species each. Dinoflagellates were represented by 75 species, with the genus *Tripos* (26 species) being the most important contributor to the community composition, followed by the genera *Protoperidinium* (19 species) and *Dinophysis* (six species).

Statistical analysis

Chlorophyll *a* shows a high positive correlation with all measured inorganic nutrients, with *r* values ranging from 0.91 ($p < 0.05$) to 0.97 ($p < 0.05$) for silicates and nitrates, respectively. Moreover, the total phytoplankton density and the diatom *C. decipiens* were positively correlated with all measured nutrients and negatively correlated with salinity (Table 2). *Skeletonema costatum* was positively correlated with nitrate, phosphate and silicate, and negatively correlated with salinity. The density of *C. affinis* was correlated with nitrite, ammonia and silicate. *Alexandrium tamarense* was positively correlated with both nitrate and ammonia (Table 2). On the other hand, *C. radiates* and *T. erythraeum* did not show any correlation with the measured parameters.

The average number of species at sites 2, 3 and 6 was 96, 90 and 85, respectively, while the lowest numbers of 19 and 30 were observed at sites 5 and 4, respectively (Fig. 5a). The values of Margalef's species richness (*d*) fluctuated between 1.08 and 6.75 at sites 5 and 2, respectively. Based on Pielou's evenness (*J'*) values, most sites showed an even distribution with the minimum value (0.15) at site 4 (Fig. 5b, c). The Shannon diversity index (*H'*) also showed a similar trend as the richness values, with a higher index observed at the sites with a larger number of species (Fig. 5d). Multidimensional scaling (MDS) based on the

Table 1

List of recorded phytoplankton species in the coastal water of Jeddah during the study period

Bacillariophyceae (Diatoms)	
Centrales	
Actinoptychus senarius (Ehrenberg) Ehrenberg, 1843	*Guinardia cylindrus* (Cleve) Hasle, 1996
Azpeitia africana (Janisch ex A.Schmidt) G.Fryxell & T.P.Watkins, 1986	*Guinardia flaccida* (Castracane) H.Peragallo, 1892
Bacteriastrum hyalinum Lauder, 1864	*Guinardia striata* (Stolterfoth) Hasle, 1996
Cerataulina dentata Hasle, 1980	*Hemialus hauckii* Grunow ex Van Heurck, 1882
Ceratulina pelagica (Cleve) Hendey, 1937	*Hemialus indicus* Karsten, 1907
Chaetoceros affinis Lauder, 1864	*Hemialus membranaceus* Cleve
Chaetoceros coarctatus Lauder, 1864	*Hemiaulus sinensis* Greville, 1865
Chaetoceros compressus Cleve, 1894	*Hemidiscus* sp.
Chaetoceros convolutus Castracane, 1886	*Leptocylindrus danicus* Cleve, 1889
Chaetoceros curvisetus Cleve, 1889	*Leptocylindrus meditarraneus* (H.Peragallo) Hasle, 1975
Chaetoceros decipiens Cleve, 1873	*Leptocylindrus minimus* Gran, 1915
Chaetoceros diadema (Ehrenberg) Gran, 1897	*Melosira* sp.
Chaetoceros didymus Ehrenberg, 1845	*Planktoniella sol* (C.G.Wallich) Schütt, 1892
Chaetoceros eibeinii Grunow, 1882	*Proboscia alata* (Brightwell) Sundström, 1986
Chaetoceros laciniosus F.Schütt, 1895	*Pseudosolenia calcaravis* (Schultze) B.G.Sundström, 1986
Chaetoceros lorenzianus Grunow, 1863	*Rhizosolenia acicularis* B.G.Sundström, 1986
Chaetoceros *mitra* (Bailey) Cleve, 1896	*Rhizosolenia acuminata* (H.Peragallo) H.Peragallo, 1907
Chaetoceros neglectus Karsten, 1905	*Rhizosolenia clevei* Ostenfeld, 1902
Chaetoceros peruvianus Brightwell, 1856	*Rhizosolenia curvata* Zacharias, 1905
Chaetoceros rostratus Ralfs, 1864	*Rhizosolenia hebetata* Bailey, 1856
Chaetoceros sp.	*Rhizosolenia imbricata* Brightwell, 1858
Chaetoceros tenuissimus Meunier, 1913	*Rhizosolenia polydactyla* Castracane, 1886
Chaetoceros teres Cleve, 1896	*Rhizosolenia robusta* G.Norman ex Ralfs, 1861
Chaetoceros tertrastichon Cleve, 1897	*Rhizosolenia setigera* Brightwell, 1858
Chaetoceros willei Gran, 1897	*Rhizosolenia simplex* Karsten, 1905
Climacodium frauenfeldianum Grunow, 1868	*Rhizosolenia* sp.
Coscinodiscus marginatus Ehrenberg, 1844	*Rhizosolenia stolterfothii* H.Peragallo, 1888
Coscinodiscus radiatus Ehrenberg, 1840	*Rhizosolenia striata* Greville, 1864
Dactyliosolen fragilissimus (Bergon) Hasle, 1996	*Rhizosolenia styliformis* T.Brightwell, 1858
Ditylum brightwellii (T.West) Grunow, 1885	*Rhizosolenia temperei* H.Peragallo, 1888
Ethmodiscus gazellae (C.Janisch ex Grunow) Hustedt, 1928	*Skeletonema costatum* (Greville) Cleve, 1873
Ethmodiscus rex (Wallich in Rattray) Hendey in Wiseman & Hendey, 1953	*Thalassiosira subtilis* (Ostenfeld) Gran, 1900
Eucampia zodiacus Ehrenberg, 1839	*Triceratium dubium* Brightwell, 1859
Pennales	
Diploneis smithii (Brébisson) Cleve, 1894	*Toxarium undulatum* Bailey, 1854
Haslea trompii (Cleve) Simonsen, 1974	*Meuniera membranacea* (Cleve) P.C.Silva, 1996
Licmophora flabellata (Grev.)C.Agardh, 1831	*Navicula directa* (W.Smith) Ralfs, 1861
Nitzschia longissima (Brébisson) Ralfs, 1861	*Thalassionema bacillare* (Heiden) Kolbe, 1955
Pleurosigma angulatum (Queckett) W.Smith, 1852	*Amphora lineolata* Ehrenberg, 1838
Pleurosigma normanii Ralfs, 1861	*Cylindrotheca closterium* (Ehrenberg) Reimann & J.C.Lewin, 1964
Pseudonitzschia australis Frenguelli, 1939	*Diploneis smithii* (Brébisson) Cleve, 1894
Pseudonitzschia inflatula (Hasle) Hasle, 1993	*Entomoneis alata* (Ehrenberg) Ehrenberg, 1845
Pseudonitzschia lineola (Cleve) Hasle, 1965	*Haslea wawrikae* (Hustedt) Simonsen, 1974
Pseudonitzschia pungens (Grunow in Cleve & Möller) Hasle, 1993	*Pseudonitzschia* turgidula (Hustedt) Hasle, 1993
Pseudonitzschia seriata (Cleve) H.Peragallo, 1899	*Licmophora juergensii** C.Agardh, 1831
Thalassionema frauenfeldii (Grunow) Tempère & Peragallo, 1910	*Lioloma elongatum* (Grunow) Hasle, 1997
Thalassionema nitzschioides (Grunow) Mereschkowsky, 1902	*Pseudonitzschia cuspidata* (Hasle) Hasle, 1993
Thallassiothrix longissima Cleve & Grunow, 1880	*Pseudonitzschia delicatissima* (Cleve) Heiden, 1928
Dinophyceae (Dinoflagellates)	
Alexandrium inseutum Balech, 1985	*Protoperidinium pallidum* (Ostenfeld, 1899) Balech, 1973
Alexandrium minutum Halim, 1960	*Protoperidinium pellucidum* Bergh, 1881
Alexandrium tamerense (Lebour, 1925) Balech, 1995	*Protoperidinium pentagonam.* (Gran, 1902) Balech, 1974
Alexandrium tamiyavanichi Balech, 1994	*Protoperidinium pyrum* (Balech) Balech 1974
Amphidinium sphenoids Wülff, 1916	*protoperidinium steinii* (Jørgensen, 1899) Balech, 1974
Amphisolenia bidentata Schröder, 1900	*Pyrocystis lunula* (Schütt) Schütt, 1896
Ceratocorys armata (Schütt) Kofoid, 1910	*Pyrocystis pseudonoctiluca* Wyville-Thompson, 1876
Ceratocorys magna Kofoid, 1910	*Scrippsiella trochoidea* (Stein) Loeblich III, 1976
Dinophysis caudata Saville-Kent, 1881	*Spiraulax jolliffei* (Murray & Whitting) Kofoid, 1911

continuation Table 1

Dinophysis doryphora (F.Stein) T.H.Abé, 1967	*Spiraulax kofoidii* H.W.Graham, 1942
Dinophysis hastata Stein, 1883	*Tripos arietinum* (Cleve) F.Gómez, 2013
Dinophysis hastata Stein, 1883	*Tripos biceps* (Claparède & Lachmann) F.Gómez, 2013
Dinophysis miles Cleve, 1900	*Tripos breve brevis* (Ostenfeld & Johannes Schmidt) F.Gómez 2013
Gonyaulux grindley Reinecke, 1967	*Tripos candelabrum* (Ehrenberg) F.Gómez, 2013
Gonyaulux polygramma Stein, 1883	*Tripos contortum* (Gourret) F.Gómez, 2013
Gonyaulux spinifera (Claparède & Lachmann) Diesing, 1866	*Tripos contrarium* (Gourret) F.Gómez, 2013
Noctiluca scintillans (Macartney) Kofoid & Swezy, 1921	Tripos declinatum (G.Karsten) F.Gómez, 2013
Ornithocercus steinii Schütt, 1900	*Tripos dens* (Ostenfeld & Johannes Schmidt) F.Gómez, 2013
Phalacroma cuneus F.Schütt, 1895	*Tripos furca* (Ehrenberg) F.Gómez, 2013
Phalacroma cuneus F.Schütt, 1895	*Tripos fusus* (Ehrenberg) F.Gómez 2013
Phalacroma mitra F.Schütt, 1895	*Tripos gibberum* (Gourret) F.Gómez 2013
Phalacroma rotundatum (Claparéde & Lachmann) Kofoid & Michener, 1911	*Tripos horridum* (Cleve) F.Gómez 2013
Podolampas bipes Stein, 1883	Tripos incisum (Karsten) F.Gómez, 2013
Prorocentrum micans Ehrenberg, 1834	*Tripos inflatum* (Kofoid) F.Gómez 2013
Protoperidinium claudicans (Paulsen, 1907) Balech, 1974	*Tripos longipes* (J.W.Bailey) F.Gómez, 2013
Protoperidinium conicum (Gran, 1900) Balech, 1974	*Tripos longissimum* (Schröder) F.Gómez, 2013
Protoperidinium crassipes (Kofoid, 1907) Balech, 1974	*Tripos lunula* (Schimper ex Karsten) F.Gómez, 2013
Protoperidinium curtipes (Jørgensen, 1912) Balech, 1974	*Tripos macroceros* (Ehrenberg) F.Gómez, 2013
Protoperidinium depressum (Bailey, 1854) Balech, 1974	*Tripos massiliense* (Gourret) F.Gómez 2013
Protoperidinium diabolus Cleve, 1900) Balech, 1974	*Tripos pentagonum* (Gourret) F.Gómez, 2013
Protoperidinium divergens (Ehrenberg, 1840) Balech, 1974	*Tripos scbrankii* (Kofoid) F.Gómez, 2013
Protoperidinium elegans (Cleve, 1900) Balech, 1974	*Tripos symmetricum* (Pavillard) F.Gómez 2013
Protoperidinium fatulipes (Kofoid, 1907) Balech, 1974	*Tripos teres* (Kofoid) F.Gómez 2013
Protoperidinium grande (Kofoid, 1907) Balech, 1974	*Tripos trichoceros* (Ehrenberg) Gómez 2013
Protoperidinium leonis (Pavillard, 1916) Balech, 1974	*Tripos vultur* (Cleve) F.Gómez, 2013
Protoperidinium oblongum (Aurivillius) Parke & Dodge, 1976	*Tripos belone* (Cleve) F.Gómez 2013
Protoperidinium oceanicum (VanHöffen, 1897) Balech, 1974	*Tripos platycornis* (Daday) F.Gómez 2013
Protoperidinium ovum (Schiller, 1911) Balech, 1974	
Cyanophyceae (blue-green algae)	
Richelia intracellularis J.Schmidt, 1901	*Trichodesmium thiebautii* Ehrenberg ex Gomont, 1892
Trichodesmium erythraeum Ehrenberg ex Gomont, 1892	
Dictyochophyceae (silicoflagellates)	
Dictyocha sp.	

Table 2

Values of Pearson's correlation coefficient (r) determined for different environmental parameters and phytoplankton as well as the dominant species

	T	S	NO$_2$	NO$_3^-$	NH$_4^+$	PO$_4^{3-}$	SiO$_4^{4-}$	Chl *a*	Total Phyto	CA	CD	CR	SC	AT	TE
T	1														
S	.043														
NO$_2$	.090	$-.820^{**}$	1												
NO$_3^-$	.050	$-.816^{**}$	$.885^{**}$	1											
NH$_4^+$	.109	$-.834^{**}$	$.998^{**}$	$.875^{**}$	1										
PO$_4^{3-}$	.056	$-.845^{**}$	$.864^{**}$	$.991^{**}$	$.855^{**}$	1									
SiO$_4^{4-}$	.029	$-.939^{**}$	$.902^{**}$	$.918^{**}$	$.909^{**}$	$.928^{**}$	1								
Chl *a*	.124	$-.791^{**}$	$.928^{**}$	$.970^{**}$	$.920^{**}$	$.958^{**}$	$.912^{**}$	1							
Total Phyto	.079	$-.822^{**}$	$.831^{**}$	$.961^{**}$	$.818^{**}$	$.984^{**}$	$.888^{**}$	$.928^{**}$	1						
CA	.307	−.364	$.571^{**}$	.349	$.582^{**}$	.333	$.440^{*}$	$.554^{**}$	.325	1					
CD	.136	$-.496^{*}$	$.820^{**}$	$.495^{*}$	$.822^{**}$	$.445^{*}$	$.534^{*}$	$.582^{**}$	.418	$.545^{*}$	1				
CR	.273	−.418	.010	.017	.060	.049	.223	−.017	.009	.058	−.061	1			
SC	−.059	$-.551^{**}$	.413	$.749^{**}$	.390	$.801^{**}$	$.611^{**}$	$.651^{**}$	$.837^{**}$	−.039	−.124	−.072	1		
AT	.248	−.296	$.543^{*}$	.308	$.550^{**}$	.289	.390	$.521^{*}$	.276	$.988^{**}$	$.538^{*}$	−.051	−.070	1	
TE	.183	.259	−.256	−.254	−.260	−.246	−.275	−.236	−.192	−.109	−.174	−.102	−.139	−.100	1

** indicates a significant correlation at 0.01 level, while * indicates a significant correlation at 0.05 level

T – temperature; S – salinity; NO$_2$ – nitrite; NO$_3$ – nitrate; NH^{4+} – ammonia; PO$_4^{3-}$ – phosphate; SiO$_4^{4-}$ – silicate; Chl *a* – chlorophyll *a*; Total Phyto – total phytoplankton; CA – *Chaetoceros affinis*; CD – *Chaetoceros decipiens*; CR – *Coscinodiscus radiates*; SC – *Skeletonema costatum*; AT – *Alexandrium tamarense*; TE – *Trichodesmium erythraeum*.

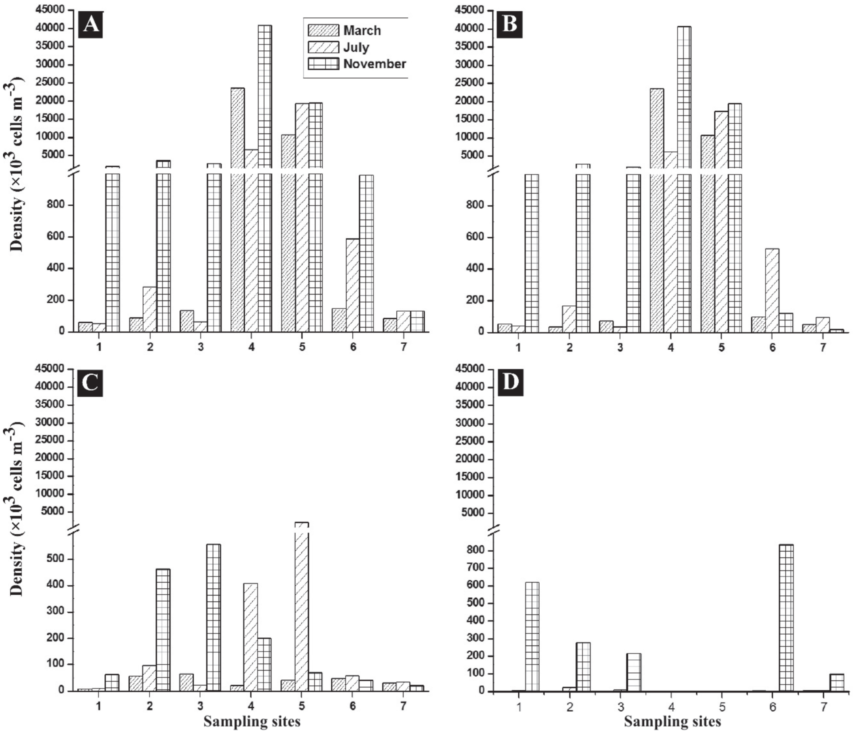

Figure 4

Densities of total phytoplankton and of specific phytoplankton groups (× 10^3 cells m^{-3}) in the study area: A) total phytoplankton, B) diatoms, C) dinoflagellates and D) cyanophytes

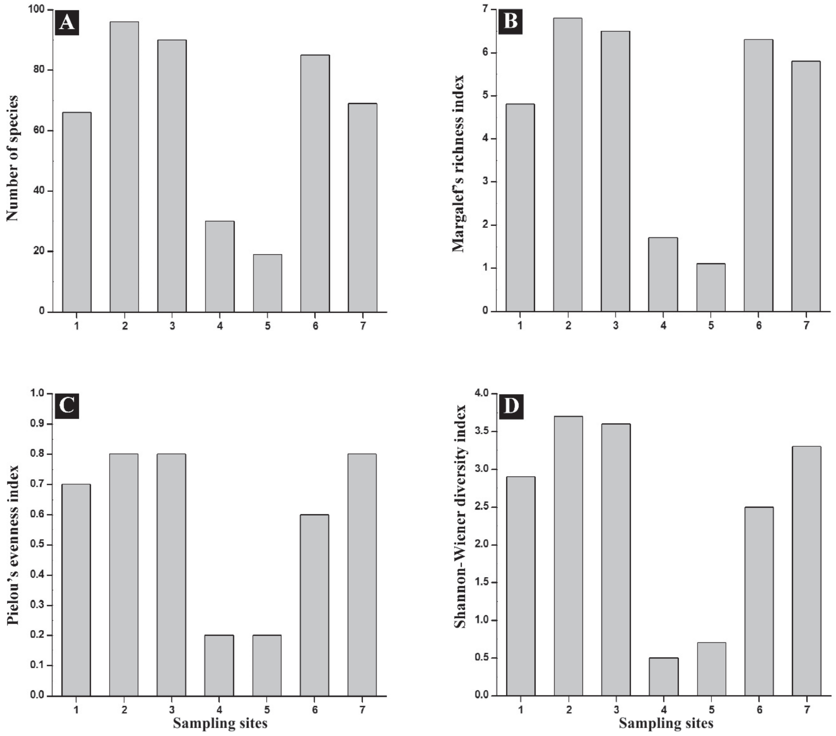

Figure 5

Diversity indices for the total phytoplankton recorded at different sites along Jeddah coastal waters: A) number of species, B) Margalef's species richness, C) Pielou's evenness index and D) Shannon–Wiener diversity index

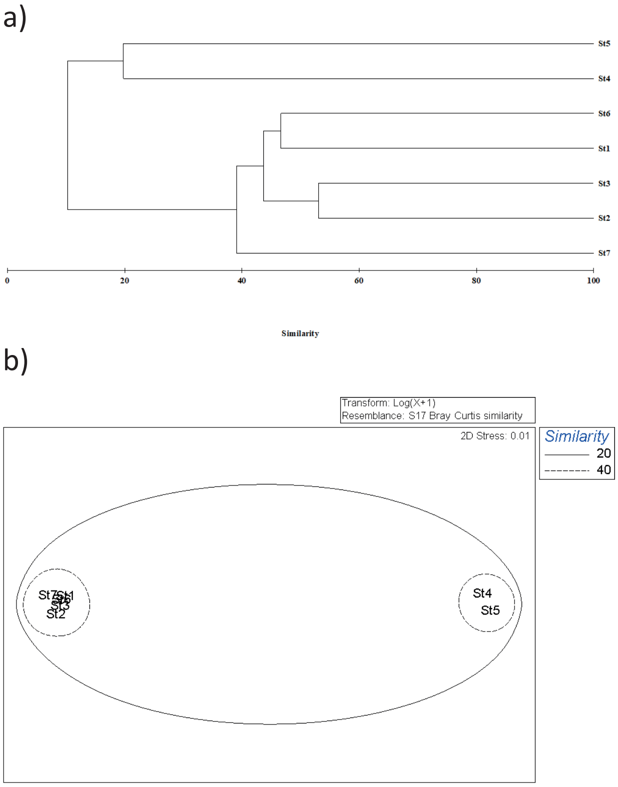

Figure 6

a) Dendrogram showing the similarity between the sites (St) based on the Bray–Curtis similarity index; b) Multidimensional scaling (MDS) ordination based on Bray–Curtis similarity indices of the total phytoplankton in the study area

Bray–Curtis similarity index of phytoplankton density showed less than 20% similarity between different sampling sites. Further, they also showed two clusters with 60% similarity (Fig. 6a, b). The results showed the presence of two main clusters, the first cluster comprises only the lagoon sites (4 and 5), which is characterized by high densities and low salinity and diversity. The other cluster comprises the neritic sites (1–3 and 6–7).

Discussion

Physicochemical parameters and phytoplankton biomass

Marine ecosystems worldwide are subjected to both natural and anthropogenic changes, which in turn affect the structure and functioning of existing communities. Rapid industrialization and increasing urbanization add more pressure to the systems by destroying the ecological balance and the food web dynamics (Maso et al. 2006). The Red Sea, known for its extreme oligotrophic characteristics, is currently changing this status at least in the lagoons and coastal waters of the growing metropolitan cities, which receive sewage effluents (Al-Aidaroos et al. 2019). In the present study, salinity showed considerable spatial variation compared to temporal variation, with low values at the lagoon sites. This can be related to the geographical location of these two sites, which receive a continuous sewage discharge of about 103 000 m^3 per day (Peña-García et al. 2014). Salinity showed a significantly high negative correlation with phytoplankton biomass, which was mainly due to the occurrence of waters with low saline at the lagoon sites. Other sites (2 and 7) receive a relatively large number of wastewater discharges and site 3 (which is located near the desalination plant), did not show any reduction in the salinity values throughout the study period. This can be associated with the mixing process as these sites and its directly connection with open waters. The present study shows abnormally high values of all measured inorganic nutrients at the lagoon sites. At these locations, the Red Sea is not oligotrophic, instead it is highly eutrophic in terms of nutrient availability. The high nutrient concentrations at the lagoon sites are in line with the previous studies (e.g. El-Rayis 1998; El-Sayed 2002a; Khomayis 2002; Al-Farawati et al. 2008; Al-Harbi & Khomayis 2010; Peña-García et al. 2014) and can be explained by the continuous inflow of sewage water into these lagoons. On the other hand, the low nutrient concentrations at other coastal locations are also comparable to studies that were carried out in different parts of the Red Sea (Dorgham et al. 2012; Kürten et al. 2015; Devassy et al. 2017, El-Sherbiny et al. 2019). As previously stated, sites 2 and 7 also received significant amounts of inorganic pollutants, but the effects were smaller. This is mainly due to the improved design of the outfall, consisting of a submerged multiport diffuser pipe dispersing the wastewater over a distance of 150 m from a water depth of 25–50 m (Peña-García et al. 2014).

In the present study, spatial differences in phytoplankton biomass values were observed, rather than the temporal ones. Chlorophyll *a* and nutrients showed a significant positive correlation ($p < 0.05$), indicating the effect of inorganic nitrogen and silicate on phytoplankton biomass. The phytoplankton biomass at the lagoon sites was very high, which was completely uncharacteristic of the typical oligotrophic nature of the Red Sea, described by Shaikh et al. (1986), Qurban et al. (2014; 2017), Kürten et al. (2015), Devassy et al. (2017) and Kheireddine et al. (2017). The chlorophyll *a* values obtained at the lagoon sites in the present study were 100–200 times higher than at the other coastal sites indicating unfavorable habitat conditions at these two sites. Stagnant conditions and continuous inflow of both treated and untreated sewage water directly toward these two sites increased the inorganic nutrient loading, which in turn allowed the available phytoplankton to proliferate and form eutrophic conditions within these lagoon locations. Sites 2, 3 and 7 are also exposed to anthropogenic impact in terms of sewage disposal, and the proximity of the desalination plant but the phytoplankton biomass was still slightly higher compared to other coastal sites.

Phytoplankton abundance and community composition

The impact of high nutrient concentrations significantly contributed to the proliferation of marine phytoplankton at some of the study sites. Traditionally known for its lower phytoplankton abundance, the Red Sea has experienced this type of phytoplankton population explosion only in a few cases. A similar type of higher phytoplankton densities was reported by Touliabah et al. (2010) and Devassy et al. (2017) from lagoon locations in the central and northern Red Sea, respectively. The densities observed at the lagoon sites during this study were much higher compared to all the studies mentioned above and can easily be compared to a tropical phytoplankton bloom. They may be related to the continuous sewage discharge into the coastal lagoons as well as their isolation from the open water. Other sites witnessed lower numbers of phytoplankton except site 2, which showed higher densities during a certain period of the study. This may be related to the anthropogenic impact mainly in the form of sewage disposal (Peña-García et al. 2014). The phytoplankton densities observed at other coastal sites are comparable to the normal pattern of phytoplankton distribution reported in the previous studies, both in the Red Sea proper and the Gulf of Aqaba (Sommer 2000; Post et al. 2002; Al-Najjar et al. 2007; Kürten et al. 2015; Devassy et al. 2017, El-Sherbiny et al. 2019).

Diatoms dominated in the phytoplankton community throughout the study period, similarly to what was described in previous studies (Shaikh et al. 1986; Devassy et al. 2017). The count of dinoflagellates was significantly low in March and showed higher values in July and November, which is consistent with the observations reported by Shaikh et al. (1986), Touliabah et al. (2010), Ismael (2015) and Kürten et al. (2015). Cyanophytes marked their presence mainly in the autumn, which is similar to the findings of Shaikh et al. (1986). The summer and autumn dominance of cyanophytes (mainly *Trichodesmium*) was reported by Post et al. (2002) from the Gulf of Aqaba and by Devassy et al. (2017) from the central Red Sea. Diatoms are known for their ability to use available inorganic nutrients and can form blooms within a short period of time. In the present study, the overall conditions were favorable for the growth of diatoms, which was reflected in their proliferation. It is known that phytoplankton biomass (rather than species composition) in the coastal waters is enriched by increasing nutrients through coastal upwelling as well as anthropogenic and riverine inputs (Polat 2007). Moreover, Officer & Ryther (1980) and Mishra et al. (2006) stated that diatom species forming a bloom require silicate as the primary nutrient and the ratio of nitrate and silicate affects the composition of phytoplankton.

Diatoms were more diverse than both dinoflagellates and cyanophytes in March and November. The number of phytoplankton species identified in the current study (174 species) was higher than that reported by Touliabah et al. (2010) – 73 species, while relatively lower than those recorded by Shaikh et al. (1986), Kürten et al. (2015) and Devassy et al. (2017). The main reason for this low diversity is the selection of sampling locations, which was primarily based on the anthropogenic impact. Typically, in such environments, phytoplankton has higher abundance but low diversity, which was the case in this study. The main diatom genera observed in the present study (*Chaetoceros* and *Rhizosolenia*) are known to be common inhabitants of the Red Sea. Similarly, dinoflagellates were dominated mainly by *Tripos* and *Protoperidinium* (Dowidar et al. 1978; Shaikh et al. 1986; Ismael 2015; Kürten et al. 2015; Devassy et al. 2017). Cyanophytes were represented mainly by *T. erythraeum*, which dominated in November. Based on the present study, the degree of human activity in Jeddah coastal waters has a significant impact on the density and diversity of phytoplankton. Furthermore, this study can be useful as a baseline for the development of monitoring programs for Jeddah coastal waters and as an aid in the implementation of a primary alert system for harmful algal blooms.

Conflict of interest

The authors declare that they have no conflict of interest.

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