Biological indicators as tools for monitoring water quality of a hot spot area on the Egyptian Mediterranean Coast

Nihal G. Shams El-Din1 · Mostafa M. El-Sheekh2 · Hala Y. El-Kassas1 · D. I. Essa1,2 · Basma A. El-Sherbiny1

Received: 24 September 2021 / Accepted: 12 August 2022 / Published online: 5 September 2022
© The Author(s) 2022

Abstract
El-Mex Bay is a semi-sheltered coastal embayment located west of Alexandria, the Egyptian Mediterranean Sea. It is considered as a hot spot area receiving industrial, agricultural, and domestic wastes from the adjacent Lake Mariut through El-Umum Drain. To evaluate the water quality of the bay, phytoplankton and macroalgae as biological tools were investigated seasonally concerning physicochemical parameters from 2018 to 2019. The results indicated that the Bay harbored a diversified phytoplankton community (162 species), comprising 99 freshwater forms, 38 marine, 22 euryhaline, and only 3 brackish forms. The total average phytoplankton abundance was 419,414 unit/L. Few species were responsible for the main bulk of phytoplankton namely Merismopedia tenuissima, Cylindrotheca closterium, Cyclotella sp., Skeletonema costatum, Scenedesmus quadricauda, Scenedesmus bijugus, and Tetradesmus dimorphus. During the study period, El-Mex Bay was affected by the presence of 13 harmful and/or toxic algal species, among which the mediophyte species Skeletonema costatum was responsible for the peak occurring during autumn (2018). The results of the recorded nutrients indicated that the Bay is highly loaded, where NH₄, NO₂, NO₃, PO₄, and SiO₄ attained 141.68 µM, 25.61 µM, 151.16 µM, 10.73 µM, and 232.86 µM, respectively. The macroalgal flora was represented only by the two opportunistic species (Ulva fasciata and Ulva intestinalis). Both species could survive in freshwater conditions exhibiting very broad salinity tolerance (6.51–38.41‰) and a high level of nitrogenous compounds. Those results revealed that El-Mex Bay suffers from pollution and deterioration of water quality.

Keywords Biological tools · Physicochemical parameters · El-Mex Bay · Mediterranean Sea

Introduction
The world’s coastal areas are of great ecological and economic importance. These areas represent natural resources, biological diversity, and a high potential for all kinds of commercial and industrial activities. They have naturally higher nutrient values than the open sea, due to inflows from land, more shallow and complex topography, higher water temperatures, and restricted water exchange (Bonsdorff et al. 2002). High population pressure due to tourism and other human activities such as fisheries and aquaculture, shipping, oil, and gas industries are leading to drastic consequences (El-Saeedy 2007; Masria et al. 2014), among which water quality degradation is the worst (El-Saeedy 2007). Pollution and eutrophication of water are major concerns on the coasts of most countries due to the discharge of several land-based effluents consisting of industrial, domestic, and agricultural wastes, non-point discharges, and discharges from shipping vessels (De Jong et al. 2002; GESAMP 1993; Masria et al. 2014). This nutrient over-enrichment impacted the aquatic ecosystems causing the degradation of water quality (Okbah et al. 2017). Marine systems eutrophication is well explained on both global and regional levels as in Jorgensen and Richardson (1996), Cloern (2001), Boesch (2002), de Jonge et al. (2002), and Wassmann and Olli (2005).

In the strict sense, eutrophication is a phenomenon that has been occurring for thousands of years. It is the process of the addition of nutrients to water bodies, leading to changes in the primary production and community composition (Shaw et al. 2003; Callisto et al. 2005). However, many adverse effects on human health have been arisen associated...
with changes in the energy flow of aquatic food webs (Smith et al. 2006; Howarth 2008; Rabalais et al. 2009), as well as economic losses (Wilson and Carpenter 1999) including the decrease in industrial and recreation uses (Le et al. 2010), losses in seafood production, and the presence of toxic phytoplankton species (Howarth 2008; Rabalais et al. 2009; Ngatia et al. 2019). The largest eutrophication-related problem in coastal areas is the excessive algal growth that can be toxic or cause serious degradation in the ecology of water bodies (Anderson et al. 2002; Bonsdorff et al. 2002). Likewise, the Mediterranean coastal areas suffer from degradation of water quality, and many hot spot areas are designated there. That is due to receiving huge volumes of wastewaters every year which are loaded by variable amounts and types of pollutants, in addition to the nitrogenous and phosphorous compounds which cause in turn a high level of eutrophication. This important problem extended to the Egyptian Mediterranean coast, resulting in elementary changes in the structure of planktonic and benthic communities and sometimes caused fish mortality. The level of eutrophication varied along the Egyptian coast according to the variations in volume and contents of discharged wastes (Dorgham 2011). Over 183 × 10⁶ m³ of untreated domestic sewage and wastewater are annually discharged into the sea through several sewers from different sources to the coast (Dorgham 2011).

El-Mex Bay, west of Alexandria, is one of the most important coastal embayment and industrial zones along the Mediterranean Sea. It is an essential fishery ground as well as a recreational area. It has been identified as one of several hot spots on the northern coast of Egypt (Khalil 1987; EEAA 2009) for its continuous increase in population, development, and environmental degradation in the last decades (Shobier et al. 2011; Abdel Ghani et al. 2013). Consequently, the bay suffers from eutrophication which was first recorded in 1985 (Dorgham 2011), with deteriorated ecological and biological conditions causing the flourish of some organisms responsible for the heavy blooms which sometimes become a regular event. Labib (1997) reported three algal blooms (9.53, 4.4, and 0.41 × 10⁶, cells/liter) during July, September and October (1993), respectively, while Mikhail (2008) recorded a red tide in May (225.5 × 10⁶, cells/liter). Both events were mainly caused by Skeletonema costatum accompanied by water discoloration, covering most of the stations of El-Mex Bay. However, intensive attention was paid to investigate physical and chemical characteristics of El-Mex Bay (e.g., Emara et al. 1984, 1992; Dorgham et al. 1987; Said et al.1991; Nessim 1994; Nessim et al. 2010; Shreadah et al. 2014; Abo-Taleb et al. 2015 and Okbah et al. 2017). Nevertheless, the previous work on the biological characteristics concerning phytoplankton standing crop as well as macroalgae was rather limited (Dorgham et al.1987; Labib 1997; Gharib 1998; Hussein and Gharib 2012).

Pollution and/or eutrophication are still major problems in El Mex Bay, with no real signs of reduced concentrations of nutrients. In this work, we evaluate the present status (April 2018- February 2019) of El-Mex Bay based on the community structure of the phytoplankton and seaweeds as biological tools concerning the physicochemical parameters. The aim is to identify gaps in the current knowledge and to provide the information necessary to control and manage El-Mex Bay as one of the hot spot areas of the Egyptian marine ecosystems successfully.

**Materials and methods**

The sampling of biological parameters (Phytoplankton and macroalgae) and the ambient seawater were performed seasonally from April (spring, 2018) to February (winter, 2019), covering five sites in El-Mex Bay (St. I, II, III, IV and V) (Fig. 1).

**Water analysis**

At each site, 1 L of seawater was collected in a polyethylene bottle for chemical analysis. Surface water temperature was measured immediately in situ using a thermometer graduated to 0.1 °C. The pH-value of water samples was measured in situ using Bench type (JENWAY, 3505 Electrochemistry Analyzer pH-meter) of ± 0.01 unit accuracy after necessary precautions in the sampling and standardization processes. Salinity was measured using the salinometer model (YSI556MPS). Dissolved oxygen was fixed in situ and measured according to the common Winkler method applied by Grasshoff (1976). The concentrations of chlorophyll-a were calculated according to Strickland and Parsons (1968). Ammonium (NH₄-N) sample was fixed in situ and was determined, using the indophenol blue technique according to Intergovernmental Oceanographic Commission (IOC 1983), while for determination of the other nutrient salts, water samples were filtered immediately using GF/C filter. The dissolved inorganic nitrogen forms (NO₂⁻ and NO₃⁻) were determined according to Strickland and Parsons (1968). Dissolved inorganic phosphorous (PO₄-P) was carried out according to Murphy and Riley (1962), which is then modified by Grasshoff et al. (1983). Silicate (SiO₂-Si) was determined by the molybdo-silicate method (Grasshoff 1964). The determination of TP and TN was carried out according to the technique described by Valderrama (1981) and modified by Koroleff (1983). The developed colors of these nutrient salts were measured spectrophotometrically using a spectrophotometer (Shimadzu UV-150–02) and the concentrations of two replicas of each nutrient salt were expressed as μM.
Biological parameters

Phytoplankton collection

A polyethylene bottle of a capacity of 1 L was fixed immediately by adding formalin (4%) for phytoplankton analysis. Estimation of the phytoplankton count was carried out by sedimentation method (Utermöhl 1958), using a binocular microscope. The results of phytoplankton count are the means of two replicas and were expressed as unit per liter (The unit comprised cells, colonies, and filaments). The identification and classification of algal taxa followed Heurk (1896), Peragallo and Peragallo (1897–1908), Lebour (1925), Cupp (1943), Hendey (1964), Sournia (1968, 1986), Dodge (1982), and Mizuno (1990). The taxonomic position of phytoplankton species was updated according to the site of AlgaeBase (Guiry and Guiry 2020).

Collection of seaweeds

Qualitative sampling Sampling of seaweeds in El-Mex Bay was done at the same five sites mentioned before from April (spring, 2018) to February (winter, 2019). Healthy specimens were collected and were washed with seawater in situ to remove the adhered sediments and impurities, then were put in labeled polyethylene bags for further identification. In the laboratory, quick rinsing of the algae with tap water was carried out on the same day to get rid of the remaining impurities and epiphytes. Identification of the two recorded species was performed according to Aleem (1993).

Quantitative sampling Samplings were conducted for the investigation of biological parameters of seaweeds which are namely species assemblages, species composition, cover %, and biomass, by applying ten replicated quadrates with 25 subdivisions and an area of 0.25 m², which were randomly taken from the littoral zone of each station at 0.5–1.0 m depth. Seaweed cover % for each subdivision in each quadrat was estimated in the field according to Saito and Atobe (1970) and Wong et al., (2012). Biomass (g fw/m²) was calculated according to Scanlan et al, (2007).

Statistical analysis

The principal component analysis was performed between the different groups of phytoplankton, harmful algae, and the physicochemical parameters. The similarity index between the sites of El-Mex Bay, based on phytoplankton community structure and their ambient water quality, was calculated by using the program Minitab 14. The species diversity (H’) was calculated according to Shannon and Weaver (1963).

For the recorded seaweeds, principal component analysis was applied between cover%, wet biomass of Ulva fasciata and Ulva intestinalis, and the physicochemical parameters of El-Mex Bay during the years 2018 and 2019.

Results

The results of the physicochemical parameters and that of the biological studies are shown in Tables 1–6 and Figs. 1–8.
Table 1  The range, average, and standard deviation of the physicochemical parameters of El-Mex Bay during (2018–2019)

| Physico-chemical parameters | Site I          | Site II         | Site III         | Site IV         | Site V          |
|-----------------------------|----------------|-----------------|------------------|----------------|----------------|
| Temperature (°C)            | 14.00–31.50    | 16.00–31.10     | 17.00–32.00      | 18.00–32.00     | 16.00–32.00     |
| pH                          | 7.73–7.90      | 7.64–7.99       | 7.58–7.82        | 7.66–8.26       | 7.62–7.92       |
| Salinity (%)                | 6.51–23.63     | 8.8–22.91       | 7.68–30.15       | 30.55–38.44     | 7.77–21.46      |
| Dissolved oxygen (mg L⁻¹)   | 3.51–9.71      | 4.21–8.42       | 3.10–9.81        | 5.88–12.72      | 3.34–10.79      |
| Chl. a (µg/l)               | 5.97 ± 2.68    | 5.70 ± 1.86     | 5.48 ± 3.03      | 8.37 ± 3.07     | 6.30 ± 3.20     |
| PO₄ (µM)                    | 0.73–9.01      | 0.57–10.73      | 0.95–4.14        | 1.82–5.77       | 3.61–21.50      |
| Nitrate (NO₃) (µM)          | 5.57–25.36     | 4.74–25.61      | 2.16–7.92        | 6.6–17.61       | 6.6–17.61       |
| Nitrite (NO₂) (µM)          | 117.82–235.34  | 130.16–207.2    | 77.59–317.23     | 17.87–107.36    | 80.62–194.15    |
| Total N (µM)                | 169.80 ± 54.61 | 167.53 ± 39.31  | 167.68 ± 103.75  | 56.24 ± 38.91   | 143.23 ± 53.43  |
| PO₄ (µM)                    | 4.21 ± 3.70    | 4.37 ± 4.54     | 2.22 ± 1.38      | 0.88 ± 0.64     | 2.55 ± 0.84     |
| Total P (µM)                | 2.07–14.88     | 2.35–14.88      | 2.18–5.32        | 0.96–5.41       | 1.46–7.83       |
| Si (µM)                     | 79.28–220.94   | 76.02–232.86    | 32.94–147.23     | 3.92–170.38     | 24.93–207.23    |
| Chlorophyll a (µg/l)        | 84.10–235.34   | 130.16–207.2    | 77.59–317.23     | 17.87–107.36    | 80.62–194.15    |
| Salinity (%)                | 6.51–23.63     | 8.8–22.91       | 7.68–30.15       | 30.55–38.44     | 7.77–21.46      |
| Dissolved oxygen (mg L⁻¹)   | 3.51–9.71      | 4.21–8.42       | 3.10–9.81        | 5.88–12.72      | 3.34–10.79      |
| Chl. a (µg/l)               | 5.97 ± 2.68    | 5.70 ± 1.86     | 5.48 ± 3.03      | 8.37 ± 3.07     | 6.30 ± 3.20     |
| PO₄ (µM)                    | 0.73–9.01      | 0.57–10.73      | 0.95–4.14        | 1.82–5.77       | 3.61–21.50      |
| Nitrate (NO₃) (µM)          | 5.57–25.36     | 4.74–25.61      | 2.16–7.92        | 6.6–17.61       | 6.6–17.61       |
| Nitrite (NO₂) (µM)          | 117.82–235.34  | 130.16–207.2    | 77.59–317.23     | 17.87–107.36    | 80.62–194.15    |
| Total N (µM)                | 169.80 ± 54.61 | 167.53 ± 39.31  | 167.68 ± 103.75  | 56.24 ± 38.91   | 143.23 ± 53.43  |
| PO₄ (µM)                    | 4.21 ± 3.70    | 4.37 ± 4.54     | 2.22 ± 1.38      | 0.88 ± 0.64     | 2.55 ± 0.84     |
| Total P (µM)                | 2.07–14.88     | 2.35–14.88      | 2.18–5.32        | 0.96–5.41       | 1.46–7.83       |
| Si (µM)                     | 79.28–220.94   | 76.02–232.86    | 32.94–147.23     | 3.92–170.38     | 24.93–207.23    |

Physicochemical parameters

The results of the physicochemical parameters of seawater samples collected from El Mex Bay during (2018–2019) are shown in Table 1. Surface water temperature in El-Mex Bay ranged from 14 °C at Site I during winter (2019) to 32 °C at Site III, IV, and V during summer (2018). The normal thermal cycle was clear in the study area, showing the highest temperature during summer (31.40 °C) against the lowest (16.20 °C) during winter (2019). The pH values were close to each other and lied on the alkaline side (7.58–8.26), with slight temporal and spatial variations. The readings of salinity were generally low at all sites, except for Site IV (30.55–38.44‰). On the temporal scale, the salinity showed the lowest value (13.68‰) during winter (2019). The content of dissolved oxygen varied spatially and temporally, attaining a maximum of 12.72 mg/L at Site IV during autumn (2018) and recording the highest average (10.29 mg/L) during this season. Levels of Chl-a in El-Mex Bay water showed great variations ranging from 0.84 (Site III) to 28.59 (Site I) µg/L, with the highest average of 17.67 µg/L at the latter site, while the seasonal highest average was recorded during winter (18.04 µg/L) (Table 1).

As far as the nutrients are concerned, a distinguishable high ammonium content was observed at all sites during the four seasons, ranging from 9.32 µM at Site IV during summer (2018) to 141.68 µM at Site I during winter (2019). Regarding nitrite, it reached a maximum value of 25.61 µM at Site III during winter (2019) against a minimum of 2.16 µM at Site IV during autumn (2018). The nitrate content was particularly high during winter (2019), attaining a maximum of 151.6 µM (Site III), with an average of 58.41 µM at this site. Total nitrogen content ranged between 17.87 µM at Site IV during autumn (2018) and 25.61 µM at Site III during winter (2019). Regarding nitrite, it reached a maximum value of 2.16 µM at Site IV during autumn (2018). The content of reactive phosphate ranged between 0.03 µM at Site IV during autumn (2018) and 14.88 µM at Site I during winter (2019). Reactive phosphate contents ranged between 0.03 µM at Site IV during autumn (2018) and 14.88 µM at Site I during winter (2019).
Table 2  Long-term study of environmental parameters in El-Mex Bay on the long-term scale

| Parameter          | After Dorgham (2011) | Labib (1997) | Soliman and Gharib (1998) | Dorgham (1997) | EEAA (2003) | EEAA (2004) | EEAA (2005) | Nessim et al., (2010) | Shreadah et al., (2014) | Okbah et al., 2017 | Present study |
|--------------------|----------------------|--------------|---------------------------|----------------|-------------|-------------|-------------|----------------------|------------------------|-----------------------|--------------|
| Study period       | 1982–1983            | 1993         | 1995                      | 1996           | 2003        | 2004        | 2005        | 2007–2008            | 2010–2011              | 2012–2013            | 2018–2019     |
| Salinity          | 5.2–38.44            | 20–39.8      | 3.68–38.5                 | 0.6–39.6       | -           | -           | -           | 7.93 – 32.3          | 28.2 – 32.3            | 7.1 – 8.7             | 6.5 – 38.4    |
| Dissolved oxygen (mg L⁻¹) | -                   | 1.6–9.8      | 0.23–8.2                  | 2.6–9.6        | -           | -           | -           | 1.96 – 5.63          | 4.95 – 6.4             | 0.00 – 3.2            | 3.1 – 12.7    |
| Nitrite (μM)      | 6.3–21.0             | 2.0–16.6     | 4.81–58.0                 | 0.0–71.0       | 2.37–12.6   | 3.97–11.5   | 5.23 – 6.0  | 13.3 – 28.9          | 6.21 – 13.1            | 0.1 – 66.7             | 1.5 – 151.2  |
| Nitrate (μM)      | 5.78–34.6            | 2.5–65.8     | 0.0–132.1                 | 2.13–127.8     | 1.5–19.6    | 1.95–5.1    | 3.29–9.4    | 31.7 – 174.4         | 70.4 – 118.7           | 4.2 – 219.1            | 9.3 – 141.7  |
| Ammonia (μM)      | -                    | -            | -                         | -              | -           | -           | -           | 77.2 – 86.3          | 99.3 – 335.1           | -                     | -            |
| Total Nitrogen (μM) | -                    | -            | -                         | -              | -           | -           | -           | 3.29–9.4             | 2.1 – 12.6             | -                     | 0.03 – 10.7  |
| Phosphate (μM)    | 4.2–19.43            | 0.6–4.4      | 0.28–17.2                 | 0.32–48.0      | 0.07–3.31   | 0.37–1.7    | 0.09–3.5    | 2.9 – 12.6           | 1.9 – 7.0              | 7.0 – 17.4             | 0.03 – 10.7  |
| Total Phosphorus (μM) | -                    | -            | -                         | -              | -           | -           | -           | -                   | 2.1 – 7.0              | 3.1 – 14.9             | -            |
| Silicate (μM)     | -                    | -            | 11.4–159.8                | -              | 2.1 – 35.2  | -           | -           | 36.3 – 149.2         | 16.8 – 48.6            | 31.7 – 78.9            | 19.1 – 232.7 |
| Chl-a (μg L⁻¹)    | 16.25–53.3           | 1.5–28.0     | 0.2–16.3                  | 3.25–50.0      | 8.33–25.2   | 1.62 – 15.9 | -           | 62.1 – 128.1         | 4.7 – 37.6             | 1.8 – 28.6             | -            |
Biological studies

Community composition

El-Mex Bay was found to be a discrete area of high phytoplankton diversity (162 species), comprising 99 freshwater forms, 38 marine, 22 euryhaline, and only 3 brackish forms. The bay was represented by nine algal groups namely Bacillariophyceae (40 species), Chlorophyceae (31 species), Cyanophyceae (23 species), Euglenophyceae (21 species), Dinophyceae (13 species), Mediophyceae (11 species), Trebouxiophyceae (10 species), Cosinodiscophyceae (9 species), and Conjugatophyceae (4 species) (Table 3). The average total count of phytoplankton amounted to 419,414 unit/L (Table 3). The Mediophyceae was the dominant group forming (43.22%) of the total count, followed by Cyanophyceae (25.12%) and Chlorophyceae (16.95%), Trebouxiophyceae (7.45%), Bacillariophyceae (4.35%) while Euglenophyceae, Cosinodiscophyceae, Dinophyceae, and Conjugatophyceae formed collectively (2.92%).

The phytoplankton diversity displayed low variations, where Site IV was the lowest diversified (96 species), while Site V was the most diversified (112 species), with only five species restricted to this site (Dinophysis caudata, Navicula anglica, Navicula rhynchocephala, Nitzschia linearis, and Protoperidinium divergens). On the other hand, there were no perceptible seasonal variations in phytoplankton diversity (92–103 species). Among the recorded species, the third was perennial (58 species) (occurring during the four seasons), against a large number occurring during one season (68 species). The rest number of the species was observed either for three or two seasons.

Table 3  The community structure and count of phytoplankton in El-Mex Bay on the long-term scale

| Study period       | Dorgham et al. (1987) | El-Sherif (1989) | Gharib (1998) | Dorgham (1997) | Present study |
|--------------------|-----------------------|------------------|---------------|----------------|---------------|
| Bacillariophyceae  | 119.00                | 83.00            | 83.00         | 41.00          | 40.00         |
| Cosinodiscophyceae | 9.00                  | -                | -             | -              | 9.00          |
| Mediophyceae       | 11.00                 | -                | -             | -              | 11.00         |
| Dinoflagellates    | 50.00                 | 5.00             | 17.00         | 14.00          | 13.00         |
| Chlorophyceae      | 26.00                 | 41.00            | 30.00         | -              | 31.00         |
| Trebouxiophyceae   | 10.00                 | -                | -             | -              | -             |
| Cyanophyceae       | 11.00                 | 26.00            | 20.00         | -              | 23.00         |
| Euglenophyceae     | 4.00                  | 4.00             | 8.00          | -              | 21.00         |
| Freshwater forms   | -                     | -                | -             | 11.00          | -             |
| Conjugatophyceae   | -                     | -                | -             | -              | 4.00          |
| Total sp. no       | 210.00                | 159.00           | 158.00        | 66.00          | 162.00        |
| Total count (×106 unit L⁻¹) | 0.04               | 0.10            | 0.94          | 31.40          | 0.42          |

Phytoplankton community

The relative abundance of the algal groups showed great variations. During spring (2018), the Mediophyceae prevailed at all sites, with the highest contribution at Site II (60.76%) against the lowest at Site IV and V (32.50 and 34.26%), respectively, forming numerically an average of 52.83% of the main bulk during this season. During summer, the three groups Cyanophyceae, Chlorophyceae, and Mediophyceae co-shared the dominance at all sites, except for Site IV, where the dinoflagellates co-dominated with the bacillariophytes and mediophytes showing a pronouncedly high percentage (41.08%). The autumn showed a distinguishable peak, where the mediophytes formed numerically the highest percentage (66.06%). This was due to its absolute dominance at Site III (85.61%) and Site IV (98.56%). During winter, the cyanophytes dominated at all sites, except for Site IV where it retrograded to 20.02% and co-shared with the bacillariophytes (26.59%), chlorophytes (24.74%), and Euglenophytes (16.82%). On the other hand, cosinodiscophytes and conjugatophytes appeared at a negligible rate during the study (Fig. 2).

Standing crop

The total abundance of phytoplankton was relatively high in the current study of El-Mex Bay (average of 419,414 unit/L) as compared with the previous studies, except for Dorgham (1997) which showed an exclusively high total count, followed by Gharib (1998) (Table 3). The total phytoplankton showed the highest count during autumn (2018), with an average of 664,319 unit/L, followed by summer (424,425 unit/L), spring (319,950 unit/L), and winter (268,964 unit/L), while on the spatial scale, the total count showed little variations (392,379–448,745 unit/L) (Table 4). However,
few species were responsible for the main bulk of phytoplankton, which are namely the cyanophyte *Merismopedia tenuissima*, the bacillariophyte *Cylindrotheca closterium*, the two mediophytes *Cyclotella* sp., *Skeletonema costatum*, and the three chlorophytes *Scenedesmus quadricauda*, *Scenedesmus bijugus*, and *Tetradesmus dimorphus*.

The high phytoplankton count (664,319 unit/L) during autumn was due to the exclusively high count of Site IV (1,374,228 unit/L) followed by Site III (593,847 unit/L). The mediophyte species *Skeletonema costatum* was responsible for the peak of autumn and dominated absolutely at Site IV (97.63%), while it contributed by 79.58% to the total count at Site III. The summer was the second season in rank. The cyanophyte *Merismopedia tenuissima* was the leader species at all sites, except for Site IV where it was completely absent giving the rank to *Cyclotella* sp. (27.86%) and the dinoflagellate *Gymnodinum variabile* (39.22%) (Fig. 3).

During spring *Cyclotella* sp. recorded the highest count at Site I, II, and III (54.92%, 59.90% and 58.31%) and co-dominated with the green algae *Scenedesmus* spp. forming collectively (16.05, 16.61, and 17.26%), respectively. The two sites (IV and V) followed the same pattern, but in addition to these species, the dinoflagellate *Prorocentrum micans* co-dominated at Site IV (8.81%), while *Merismopedia*...
tranquilla ranked in the second-order (16.35%) at Site V (Fig. 3).

The winter season sustained the lowest count (268,964 unit/L) during the study. Merismopedia spp. were dominant, attaining the highest percentage (50.62%) at Site II against their lowest percentage (19.75%) at Site IV. The mediophyte Cyclotella sp. and the diatomate Cylindrotheca closterium contributed with (3.32–9.60%) and (8.12–11.10%) to the total count, respectively, while Scenedesmus spp. co-shared at all the sites (Fig. 3).

During the study period, 13 harmful and/or toxic algal species were recorded throughout the Bay, among which seven species belong to the Cyanophyceae (Anabeanopsis circularis, Merismopedia tenuissima, Microcystis aeruginosa, Kamptonema formosum, Oscillatoria limnetica, Oscillatoria tenuis, and Planktothrix agardhii), in addition to two bacillariophytes (Cylindrotheca closterium and Pseudo-nitzschia sp.), two mediophytes (Chaetoceros affinis and Skeletonema costatum), and two dinoflagellates (Dinophysis caudata and Prorocentrum micans) (Table 5). These species almost contributed with relatively low total count to the main bulk, except for Cylindrotheca closterium, Merismopedia tenuissima, and Skeletonema costatum. The two former species attained a maximum of 36,100 and 171,200 unit/L, respectively during winter, while the latter species showed a pronouncedly high count (1,341,700 unit/L) during autumn (Table 5).

**Diversity index**

Absolute values of the diversity index showed wide temporal and spatial variations from the minimal diversity index of 0.18 nats recorded during autumn at Site IV and...
the maximum of 3.31 nats during winter at the same site (Fig. 4). On the temporal scale, winter (2019) sustained the highest average diversity (2.57 nats), while autumn (2018) recorded the lowest value (1.63 nats). On the other hand, all the sites recorded close average values of diversity index, except for Site V (2.53 nats).

**Statistical analysis**

The statistical analysis of the data revealed that the Trebouxiiophyceae and bacillariophytes were mainly influenced by silicate. On the other hand, the Cyanophyceae and Chlorophyceae were the most influenced groups by the physicochemical parameters. Dinophyceae and Mediophyceae were positively affected by pH, dissolved oxygen, temperature, and salinity, while they were negatively affected by ammonium and nitrite. The Conjugatophyceae

---

**Table 5** Harmful and toxic algae (unit/liter) recorded in El-Mex bay during 2018–2019

| Species                                      | Spring 2018 | Summer 2018 | Autumn 2018 | Winter 2019 | Toxins                                      | References                        |
|----------------------------------------------|-------------|-------------|-------------|-------------|--------------------------------------------|-----------------------------------|
| *Anabaenopsis circula-ris* (G.S.West) Woloszynska, V.Y.Miller | 0–87        | 92–593      | 0–18        | 0–44        | Microcystin                                | Coulibaly et al. (2014)           |
| *Chaetoceros affinis* Lauder                 | 0–2400      | 0           | 0–5400      | 0           | Bloom-forming—kill fish                    | Lim et al. (2014)                 |
| *Cylindrotheca closterium* (Ehrenberg) Reimann & J.C.Lewin | 3200–5400   | 200–9400    | 800–3300    | 8300–36,100 | Water discoloration & Potentially harmful | Mikhail (1997) Wang et al. (2008) |
| *Dinophysis caudata* W.S. Kent                | 0–100       | 0           | 0           | 0           | Pectenotoxin-2 (Lipophilic, diarrhetic Shellfish Toxins) | Basti et al. (2015) Fernández et al. (2019) |
| *Kamptonema formosum* (Bory ex Gomont) Strunecký, Komárek & J. Šmarda | 0–829       | 0–192       | 0           | 0–48        | Microcystins, homoanatoxin-a               | Carmichael (2013)                 |
| *Merismopedia tenuissima* Lemmermann         | 0–12,000    | 0–150,200   | 3200–193,200| 13,600–171,200| Nodularin                                   | Jakubowska and Szlag-Wasielewska (2015) |
| *Microcystis aerigunosa* (Kützing) Kützing    | 0           | 0           | 0           | 0–200       | PTS, Microcystins                          | Moura et al. (2018)               |
| *Planktothrix agardhii* (Gomont) Anagnos-tidis & Komárek | 0–2791      | 0–1736      | 70–933      | 175–1954    | Microcystins                               | Tonk et al. (2005)                |
| *Pseudo-nitzschia sp.*                       | 0–1500      | 0–200       | 0–1100      | 100–400     | Domoic acid                                | Orsini et al (2002)               |
| *Oscillatoria linnetica* (Lemmermann) Komárek | 0–384       | 0–1082      | 0–428       | 0–3968      | Microcystins                               | Mohamed (2016)                    |
| *Oscillatoria tenuis* C.Agardh ex Gomont      | 0–445       | 0–2982      | 0–1509      | 0–87        | Microcystins                               | Carmichael (2013)                 |
| *Prorocentrum micans* Ehrenberg               | 300–9050    | 0–300       | 0–100       | 0–800       | High biomass harmful                       | Ignatiades and Gotsis-Skretas (2010) |
| *Skeletonema costatum* (Greville) Cleve       | 0           | 200–31,100  | 13,700–1,341,700 | 0–1500 | Brown water, water coloration, clogging fish gills & arresting the development of invertebrate eggs | Gastrich (2000) Mendez and Ferrari (2002) Malone (2007) D’Ippolito et al. (2002)|

---

![Fig. 4](image-url) Seasonal variations of species diversity at the different stations in El-Mex Bay during 2018–2019
and Euglenophyceae were negatively influenced by the temperature and were positively affected by nitrate and total nitrogen. The cosinophytes were mainly affected by total nitrogen and nitrate (Fig. 5). For harmful algae, the principal component analysis between the abundance of each species and the physicochemical parameters showed different relationships (Fig. 6). The similarity index revealed three clusters (Fig. 7).

**Macroalgal abundance at El-Mex Bay during 2018–2019**

El-Mex Bay has a physiognomic monotony where the algal flora was represented by only two chlorophyte species (*Ulva fasciata* Delile and *Ulva intestinalis* Linnaeus), mostly alternating each other in the different sites. The former species was the less abundant, appearing only in...
sites III and IV, while the latter species was more abundant, appearing in four sites (I, II, III, and V). For *U. fasciata*, the average cover was (14.05 and 33.27%) at Site III and IV respectively, and formed the lowest total average (9.46%), while the average cover of *U. intestinalis* ranged from (12.91%) at Site V to (19.46%) at Site I (Table 6), with total average cover (13.49%). On the temporal scale, the cover of *U. fasciata* was relatively low and did not exceed 9%, except during winter (2019), it attained the double. For *U. intestinalis*, it attained the highest cover (16.59%) during autumn (2018) (Table 6).

The wet biomass of *U. fasciata* at sites III and IV attained an average of 137.61 and 354.37 g FW/m², respectively (Table 6), with a total average of 98.4 g FW/m². The average wet biomass of *U. intestinalis* ranged from 49.59 (Site I) to 155.75 g FW/m² (Site III) (Table 6), with a total average of 77.72 g FW/m². On the temporal scale, the biomass of *U. fasciata* was low during summer, with an average of 77.63 g FW/m², while it attained high values of 135.11 and 136.30 g FW/m² during spring and winter, respectively. For *U. intestinalis*, it attained the highest value (149.72 g FW/m²) during autumn (2018), against the lowest biomass during winter (2019) (23.31 g FW/m²) (Table 6).

The results of the principal component analysis between the abundance of *Ulva* spp. (cover and wet biomass) and the physicochemical parameters of El-Mex Bay during (2018–2019) revealed that the cover and wet biomass of *U. fasciata* was mainly influenced negatively by silicate followed by phosphate and total phosphorus, while the abundance of *U. intestinalis* showed the inverse pattern (Fig. 8).

**Discussion**

One of the major environmental problems in Alexandria city is seawater pollution, where various pollutants are dumped daily by industrial, agricultural, and domestic sources over Alexandria coasts through several outfalls. El-Mex Bay is one of these disposal sites (Abdallah 2007). It is a highly dynamic system subjected to a high allochthonous injection leading to continuous replenishment of nutrients throughout the year accompanied by the exchange of water with the open sea. Manifestation and symptoms of nutrient enrichment in El-Mex Bay impacted the ecological conditions of the Bay and caused pronounced drastic environmental changes, consequently affecting the biota (Labib 1997).

On a global scale, the temperature is considered an important explanatory variable for the rate of oxygen consumption, phytoplankton nutrient uptake (López-Urrutia and Morán 2015), metabolism, growth rates, survival, distribution, biomass (Chen et al. 2017; Okbah et al. 2017), and community structure (Edwards et al. 2016; Zalat et al. 2021). In the present study, the high temperature limited the growth of Euglenophyceae and Conjugatophyceae, while it enhanced the Dinophyceae and Mediophyceae. The temperature range was compatible with that recorded by Okbah et al. (2013) and Zakaria et al. (2007) which were 18–31 and 14.5–31.0 °C in El-Mex Bay water, respectively. Changes in pH can indirectly affect phytoplankton by altering the dissolved metal concentrations so that timing and abundance of species (Granéli and Haraldsson 1993). Dissolved oxygen is a key descriptor
Table 6  Cover % (species/m²) and wet biomass (g FW/m²) of *U*/*ha* spp. at the five sites of El- Mex Bay during 2018–2019

| Species/Staion          | Site I | Site II | Site III | Site IV | Site V | Site I | Site II | Site III | Site IV | Site V | Seasonal average |
|-------------------------|--------|---------|----------|---------|--------|--------|---------|----------|---------|--------|-----------------|
|                         | Cover  | Biomass |          |         |        |        |         |          |         |        |                 |
| *U. fasciata*           | 8.31   | 14.29   | 9.92     | 0       | 38.91  | 27.02  | 101.48  | 78.6     | 0       | 14.29  | 51.78           |
| *U. intestinalis*       | 17.67  | 16.26   | 22.20    | 0       | 0      | 87.94  | 116.46  | 200.02   | 0       | 0      | 11.23 101.11    |
|                         | 0      | 0       | 0        | 25.56   | 0      | 0      | 0       | 0        | 310.51  | 0      | 5.11  77.63     |
| *U. intestinalis*       | 13.61  | 28.56   | 40.03    | 0       | 0.75   | 67.48  | 234.72  | 344.38   | 0       | 0      | 16.59  149.72   |
|                         | 0      | 0       | 0        | 44.55   | 0      | 0      | 0       | 0        | 435.48  | 0      | 8.91  87.10     |
| *U. fasciata*           | 38.25  | 9.04    | 0        | 0       | 11.98  | 15.89  | 67.6    | 0        | 0       | 0      | 33.04  11.85    |
| *U. intestinalis*       | 0      | 0       | 14.05    | 33.27   | 0      | 0      | 0       | 137.61   | 354.37  | 0      | 18.00  23.31    |
| Average of sites (U.   | 0      | 0       | 14.05    | 33.27   | 0      | 0      | 0       | 137.61   | 354.37  | 0      | 18.00  23.31    |
| Average of sites (U.    | 19.46  | 17.04   | 18.04    | 0       | 12.91  | 49.59  | 130.07  | 155.75   | 0       | 0      | 18.00  23.31    |

**Species/Station**

| Spring 2018 | Summer 2018 | Autumn 2018 | Winter 2019 |

Average of sites (U. *fasciata*): 19.46
Average of sites (U. *intestinalis*): 17.04
of water quality in coastal areas where prolonged changes in oxygen levels could lead to modification in the local biotic community structure (Jack et al. 2009). In general, for a diversified warm-water biota, the optimum concentrations should not be less than 4.65 mgO2/L, while for a cold-water biota, concentrations should not be lower than 5.59 mgO2/L for desirable healthy growth (Grundy 1971; Arin 1974). Referring to Stachowitsch and Avcin (1988), the oxygen concentration < 2 mg/l reflects hypoxia and negatively impacts marine organisms. Accordingly, the Bay was well oxygenated reaching 12.72 mg/L at Site IV during autumn which exceeded the ranges of all the previous studies. On the spatial scale, the distribution of surface salinity throughout El-Mex Bay demonstrated wide-range variations relative to the dispersal pattern of wastewaters discharged into the Bay. Near-shore waters salinity sustained usually low values and increasing seaward to exceed 30‰ in the open part of the Bay (Dorgham 2011). Based on the surface salinity in El-Mex Bay, four water masses overlaying each other could be identified, where the area and position of each mass are directly governed by wind direction showing a seasonal pattern. Water masses include (1) mixed land drainage of a salinity < 10.00 ppt, (2) mixed water (10.0–30.0 ppt), (3) diluted seawater (30.0–38.5 ppt), and (4) Mediterranean seawater (> 38.50 ppt) (Soliman and Gharib 1998; Zakaria et al. 2007). In the current study, the salinity at all sites falls between mixed land drainage salinity and mixed water salinity, except for Site IV which followed the diluted seawater salinity regime. On the other hand, El-Sherif (1989) reported that the drain water flows as a surface current into the Bay along the western and the northwest coasts during the autumn, where a dense water mass with a high phytoplankton count occupies the center of the bay, associated with an increase in salinity during this season. Our results agreed with El-Sherif (1989), where a dense water mass with a high phytoplankton count was recorded at Site IV during autumn, with salinity (30.55–38.44‰).

In natural waters, nutrient salts are considered essential compounds for living organisms in aquatic systems. They represent the fertility of water, on which primary productivity and, eventually fish production depend. The large content of phosphate is considered a potential pollutant due to the huge load of domestic and drainage effluents discharging into El-Mex Bay (Said et al. 1994; Fahmy et al. 1997). PO4 concentration ranged from 0.03 to 10.73 μM/L in El-Mex Bay during the study period, which was higher than that of Labib (1997) (0.6–4.4 μM/L), Shreadah et al. (2014) (1.93–7.01 μM/L) and particularly EEAA (2003, 2004, 2005) and exceeded the criteria of Stirn (1988) for polluted water (0.3–0.5 μM/L). Due to the high phosphate content in El-Mex Bay during the study, this salt was not a limiting factor for the growth of phytoplankton groups, except for Cyanophyceae and Chlorophyceae. The nitrogenous compounds are the most significant indicators of water quality. The concentrations of ammonium (9.32–141.68 μM) and nitrate (1.48 to 151.6 μM) in El-Mex Bay were excessively high, compared with the previous studies and exceeded that adopted by Vucak and Stirn (1982) for eutrophic areas (> 2.0 μM/L NH4 and > 4.0 μM/L NO3). Silicate is another important parameter controlling marine productivity and is considered a good indicator of freshwater dispersion. It is potential for diatom blooms (Okbah et al. 2017). In this study, the bacillariophytes were antagonized with silicate concentration.
This may be due to the uptake of silicate by this group (previously included in diatoms class) during their growth (Dorgham et al. 2004). On the other hand, the range of silicate (3.92–232.86 µM/L) highly exceeded that of all previous studies. However, coastal waters have, in general, quite high silicate contents since they are affected by land-based sources (Riley and Chester 1971). Chlorophyll-a concentration reached a maximum of 28.59 during winter at Site I. This agreed with the records over the past three decades in the Bay that indicated pronouncedly high chlorophyll-a concentrations in the bay as reported by Dorgham (2011), who classified the Bay as eutrophicated based on these concentrations and the high nutrient levels. In the current study, we can classify the Bay as a nutrient over-enriched area due to the high level of nutrients, but well-oxygenated sites.

The phytoplankton community of El-Mex Bay consisted mainly of a high number of fresh and brackish water species distributed all over the bay (102 species). Those species can tolerate the salinity gradient, prevailing in the Bay. During the present study, the phytoplankton abundance attained an average of 0.419×10^6 unit/L. However, this value was lower than that recorded by Gharib (1998) (0.940×10^6 unit/L) and the abnormal increase reported by Dorgham (1997) (3.14×10^4 unit/L) while, they were higher than that of Dorgham et al. (1987) and El-Sherif (1989) (0.043 and 0.097×10^6 unit/L), respectively. The difference in the phytoplankton abundance can be attributed to the variability of ecological conditions prevailing in the Bay.

The similarity index of the different sites in El-Mex Bay (2018-2019) based on phytoplankton abundance and physicochemical parameters showed three clusters. The first and second clusters consisted of St. IV and III respectively, while the third one included St. I, II, and V. The first two clusters that include one site each reflected the difference in the phytoplankton community structure and the physicochemical parameters than the other three close stations.

The composition, abundance, and growth of phytoplankton species are affected by a variety of environmental variables such as salinity, nutrients, and temperature (Dayala et al. 2014). As all marine coastal areas, Mediophyceae (previously included in the diatoms group in the preceding classification) demonstrated the most effective contribution to the total phytoplankton count. They are perennial forms with short generation time (Goma et al. 2005), usually producing great successive blooming pulses (Labib 2002). The prevalence of Mediophyceae might be due to their ability to adapt quickly to changes in hydrographical conditions (Rajesh et al. 2001) or attributed to the expected high silicate content in the outlet of El-Umoum drain (Dorgham et al. 1987). Their predominance over different groups is similar to observations in many other eutrophic Egyptian Mediterranean water, for example, Eastern Harbor, Kayet Bey (Labib 2002; Khairy et al. 2014), in Rosetta and Abu-Qir (Shams El Din et al. 2014). It is worth mentioning that the co-sharing of Chlorophyceae, Cyanophyceae, and Euglenophytes may be attributed to the allochthonous freshwater forms with the high load of nutrients. This was confirmed by the negative relationship between cell densities of Cyanophyceae and Chlorophyceae and salinity.

Although El-Mex Bay is highly polluted, the diversity experienced mostly high values characteristic of healthy marine ecosystems (close to 3 or >3) (Whilm and Doris 1966), except for St. IV (0.18 nats) during autumn due to the absolute dominance of Skeletonema costatum.

During this study, the community harbored few dominant species during the different seasons. The peak of autumn was mainly due to the overwhelming of the neritic mediophyte, S. costatum contributing by 61.26% to total abundance. It is a cosmopolitan and familiar red tide causative species (Mingazzini et al. 1992) and is considered an eutrophication indicator worldwide (Pucher-Petkovic and Marasović 1980; Miñnea 1985; Revelante and Gilmartin 1985; Toming and Jaanus 2007) and was responsible for brown water coloration (Gastrich 2000; Mendez and Ferrari 2002), clogging fish gills (Malone 2007), and arresting the development of invertebrate eggs (D’Ippolito et al. 2002). Its flourishing coincided with high salinity, low levels of nitrogenous compounds, which was consistent with the principle component analysis. Dorgham (1997) reported a red tide in El-Mex Bay during the early summer with an average count of 118.5×10^6 unit/L with an absolute dominance of S. costatum. Furthermore, the species caused two red tide events (Labib 1997; Mikhail 2008) accompanied by water discoloration. The second dominant species Cylindrotheca closterium is a pollution indicator (Ignatiades and Pagou 1985) and potentially harmful, causing water discoloration (Wang et al. 2008). It was prominent at all sites during winter and was negatively influenced by temperature and positively affected by nitrogenous compounds. These results were supported by the principal component analysis. Cyclotella sp. was responsible for the peak during spring. It showed only an inverse relationship with nitrate (r = −0.356). The species was reported as the most dominant mediophyte in the Bay (El-Sherif 1989). The genus Cyclotella is of wide distribution in the Egyptian Mediterranean Coast (Gergis 1983) in addition to the Egyptian Delta lakes (Samaan 1974). This may be due to that it is the euryhaline genus (Roubeix and Lancelot 2008). Furthermore, Palmer (1969) classified the genus Cyclotella among the organic pollution indicators. Summer and winter high density were attributed to the
proliferation of *Merismopedia tenuissima* that exhibited close association with high nutrient concentrations, while it was negatively affected by salinity which was confirmed by the principal component analysis.

During the study period in El-Mex Bay, several harmful and/or toxic phytoplankton species were detected, mostly were cyanophytes which are microcystin producers. This toxin causes lethal poisoning and disturbances in the main development process, especially the initial life phases of many planktonic and fish species (Zanchett and Oliveira-Filho 2013). For a human being, it is hepatotoxic, inhibiting protein phosphatasates, genotoxic, neurotoxic, and tumorgenic (Turner et al. 2018). The diatomate *Pseudo-nitzschia* sp. produces domoic acid which is a potent exotoxin linked to marine mammal mortalities, while the mediophyte *Chaetoceros affinis* can be responsible for fish mortality. The dinoflagellates *Dinophysis caudata* and *Prorocentrum micans* were found to be harmful to autochtonous species (Zaghloul 1994). Although the total count of most of these harmful species did not exceed 100 cells/L, they were considered dangerous as some algal toxins are extremely potent (Turner et al. 2018). However, they could be considered as a symptom of eutrophication as reported by previous studies (Madkour et al. 2007).

**Macroalgae**

The use of bioindicators to monitor coastal pollution has advantages over the physicochemical ones as they are supposed to be a direct measure of pollution influence on the aquatic ecosystem (Wan et al. 2017). The ecological importance and distinguishing features of macroalgae, particularly as indicators of nutrient pollution, make them favorable for evaluation endpoints for numeric nutrient criteria development for water quality management purposes (EPA 2014).

El-Mex Bay had not been investigated before for macroalgal distribution in the previous studies. During the present study, the algal flora was represented only by two green algal species namely *Ulva fasciata* and *Ulva intestinalis*. The *Ulva* spp. are opportunistic species that can thrive causing ecological problems due to the formation of persistent blooms which may be enhanced by anthropogenic nutrient inputs from industrial, agricultural, and domestic wastes (Guidone and Thornber 2013; Smiatek and Zingone 2013). The *U. fasciata* was restricted to Sites III and IV accompanied with the relatively low level of silicate with total average (87.35 and 60.62 μM), phosphate (2.22 and 0.88 μM), and total phosphorus (4.27 and 2.45 μM) in these sites respectively, while *U. intestinalis* showed inverse pattern thriving in the sites associated with a high level of these nutrients. This could explain their distribution alternatively. On the other hand, both could survive in freshwater conditions exhibiting very broad salinity tolerance (6.51–38.41%) (Edwards et al. 1988; Cohen and Fong 2004) and a high level of nitrogenous compounds. The presence of these species, particularly *U. intestinalis*, let to suggest that El-Mex Bay suffers from pollution and deterioration of water quality.

**Conclusion**

El-Mex Bay suffers from the continuous supply of nutrients that surpassed the permissible values, causing deterioration in water quality to a non-favorable degree for healthy populations of algal flora. This was reflected by the presence of 13 harmful and/or toxic algal species that can cause serious problems to human health. In addition, the presence of two opportunistic algal species (*Ulva fasciata* and *U. intestinalis*) in the bay is an indicator of nutrient enrichment. Pollution recovery of the Bay is a must and can be performed by different treatments, among which the biological one is the best.

**Funding** Open access funding provided by The Science, Technology & Innovation Funding Authority (STDF) in cooperation with The Egyptian Knowledge Bank (EKB).

**Declarations**

All authors state that there are no ethical statements contained in the manuscripts.

**Conflict of interest** The authors declare no competing interests.

**Open Access** This article is licensed under a Creative Commons Attribution 4.0 International License, which permits use, sharing, adaptation, distribution and reproduction in any medium or format, as long as you give appropriate credit to the original author(s) and the source, provide a link to the Creative Commons licence, and indicate if changes were made. The images or other third party material in this article are included in the article's Creative Commons licence, unless indicated otherwise in a credit line to the material. If material is not included in the article's Creative Commons licence and your intended use is not permitted by statutory regulation or exceeds the permitted use, you will need to obtain permission directly from the copyright holder. To view a copy of this licence, visit http://creativecommons.org/licenses/by/4.0/.

**References**

Abdullah MAM (2007) Speciation of trace metals in coastal sediments of El-Mex bay south Mediterranean Sea-West of Alexandria (Egypt). Environ Monit Assess 132(1–3):111–123. https://doi.org/10.1007/s10661-006-9507-z

Abdel Ghani SA, Shobier AH, Shreadah MA (2013) Assessment of arsenic and vanadium pollution in surface sediments of the Egyptian Mediterranean coast. Int J Environ Tech Manag 16(1–2):82–101. https://doi.org/10.1504/ijetm.2013.050673

Abo-Taleb HA, El Raey M, Abou Zaid MM, Aboul Ezz SM, Abdel Aziz NE (2015) Study of the physico-chemical conditions and evaluation of the changes in eutrophication-related problems in El-Max Bay. Afr J Environ Sci Technol 9(4):354–364. https://doi.org/10.1016/j.ejar.2014.05.001
GESAMP (IMO/FAO/UNESCO/WMO/IAEA/UN/UNEP); Joint Group Experts on the Scientific Aspects of Marine Pollution (1993) Impacts of oil and released chemicals and wastes on the marine environment: Reports and studies No.50, International Maritime Organization, London.

Gharib SM (1998) Phytoplankton community structure in Mex bay, Alexandria, Egypt. EJABF 2(3):81–104. https://doi.org/10.21608/ejabf.1998.1632

Goma J, Rimet F, Cambra J, Hoffmann L, Ector L (2005) Diatom communities and water quality assessment in Mountain Rivers of the upper Segre basin (La Cerdanya, Oriental Pyrenees). Hydrobiologia 551(1):209–225. https://doi.org/10.1007/s10750-005-4462-1

Graneli E, Haraldsson C (1993) Can increased leaching of trace metals from acidified areas influence phytoplankton growth in coastal waters? Ambio 22(5):308–311

Grasshoff K (1964) On the determination of silica in sea water. Anal Chim Acta 33(9):127–134. https://doi.org/10.1016/S0003-2670(00)84243-7

Grasshoff K (1971) Strategies for control of man-made eutrophication. Ambio 20(4):149–156. https://doi.org/10.1007/BF00651862

Grasshoff K, Ehrhardt M, Kremling K (1983) Methods of seawater analysis. Verlag Chemie, Weinheim, p 419. https://doi.org/10.1002/9781118824618

Grasshoff K, Ehrhardt M, Kremling K (1983) Methods of seawater analysis. Verlag Chemie, Weinheim, p 317

Grasshoff K (1976) Methods of seawater analysis. Verlag Chemie, Weinheim, p 317

Grasshoff K (1976) Methods of seawater analysis. Verlag Chemie, Weinheim, p 317

Grasshoff K (1964) On the determination of silica in sea water. Anal Chim Acta 33(9):127–134. https://doi.org/10.1016/S0003-2670(00)84243-7

Grasshoff K (1971) Strategies for control of man-made eutrophication. Ambio 20(4):149–156. https://doi.org/10.1007/BF00651862

Grasshoff K, Ehrhardt M, Kremling K (1983) Methods of seawater analysis. Verlag Chemie, Weinheim, p 419. https://doi.org/10.1002/9781118824618

Grasshoff K, Ehrhardt M, Kremling K (1983) Methods of seawater analysis. Verlag Chemie, Weinheim, p 317

Grasshoff K (1976) Methods of seawater analysis. Verlag Chemie, Weinheim, p 317

Guidone M, Thornber CS (2013) Examination of U/la bloom species richness and relative abundance reveals two cryptically co-occurring bloom species in Narragansett Bay, Rhode Island. Harmful Algae 24:1–9. https://doi.org/10.1016/j.hal.2012.12.007

Guiry MD, Guiry GM (2020) AlgaeBase. World-wide electronic publication, National University of Ireland, Galway. http://www.algaebase.org

Hendey NI (1964) An introductory account of the smaller algae of British coastal waters. Part 5, Bacillariophyceae (Diatoms) Fishery Investigation, Series IV. Her Majesty’s Stationery Office, London, p 317

Heurk VH (1896) A treatise on the Diatomaceae. Translated by WE Baxter, William Wesley and Son, London p 558

Howarth RW (2008) Coastal nitrogen pollution: a review of sources and trends globally and regionally. Harmful Algae 8:14–20

Hussein NR, Gharib SM (2012) Studies on spatio-temporal dynam-ics of phytoplankton in the Merambong Shoal, Tebrau Straits with note on potentially harmful species. Malay Nat J 66(1–2):198–211

López-Urrutia A, Morán XAG (2015) Temperature affects the size-structure of phytoplankton communities in the ocean. Limnol Oceanogr 60(3):733–738. https://doi.org/10.1002/lno.10049

Madkour F, Dorgham M, Fahmy M (2007) Short term scale obser-vations on phytoplankton in the Eastern Harbor of Alexandria, Egypt. Egypt J Aquat Res 33(1):103–209

Malone TC (2007) Ecosystem dynamics, harmful algal blooms and operational oceanography. Oceanography and Stewardship of marine ecosystems 527–560

Masria A, Negm A, Iskander M, Saavedra O (2014) Coastal zone issues: a case study (Egypt). Procedia Eng 70:1102–1111. https://doi.org/10.1016/j.proeng.2014.02.122

Mendez S, Ferrari G (2002) Harmful algal blooms in Uruguay: anteced-ents, ongoing projects and review of results. In Sar EA, Ferrario ME, Regueira B (Eds) Floraes algales nocivas en el Cono Sur Americano, (In Spanish), pp 271–288

Mihnea PE (1985) Effect of pollution on phytoplankton species. Rapp Comm Int Mer Medit 29(9):85–88. https://publons.com/journal/115376/rapp-comm-int-mer-medit/

Mikhail SK (1997) Ecological Studies of the Phytoplankton in Mex Bay. PhD Thesis. Alexandria University, Egypt pp 266

Mikhail SK (2008) Dynamics of estuarine phytoplankton assemblages in Mex Bay, Alexandria (Egypt): Influence of salinity gradients. EJABF 12(4):231–251. https://doi.org/10.21608/ ejabf.2008.2014

Mingazzini M, Rinaldi A, Montanari G (1992) Multi-level nutrient enrich-ment bioassays on Northern Adriatic coastal waters. In: Grasshoff K, Eberhardt M, Kremling K (eds) Methods of seawater analysis. Verlag Chemie, Weinheim, pp 205–206

Labib W (1997) Eutrophication in Mex Bay (Alexandria, Egypt), Environmental Studies and Statistical Approach. Bull Nat Inst Oceanogr Fish A.R. Egypt 23:49–68

Labib W (2002) Phytoplankton variability in the Eastern Harbour Alex-andria (Egypt). EJABF 6(2):75–102. https://doi.org/10.21608/ ejabf.2002.1741

Le C, Zha Y, Li Y, Sun D, Lu H, Yin B (2010) Eutrophication of Lake waters in China: cost, causes, and control. Environ Manage 45:662–668

Lebour MV (1925) The dinoflagellates of the northern seas. Plymouth U.K.: Marine Biological Association of the United Kingdom pp 172

Lim HC, Teng ST, Leap CW, Iwataki M, Lim PT (2014) Phyto-plankton assemblage of the Merambong Shoal, Tebrau Straits with note on potentially harmful species. Malay Nat J 66(1–2):198–211

López-Urrutia A, Morán XAG (2015) Temperature affects the size-structure of phytoplankton communities in the ocean. Limnol Oceanogr 60(3):733–738. https://doi.org/10.1002/lno.10049

Madkour F, Dorgham M, Fahmy M (2007) Short term scale obser-vations on phytoplankton in the Eastern Harbor of Alexandria, Egypt. Egypt J Aquat Res 33(1):103–209

Malone TC (2007) Ecosystem dynamics, harmful algal blooms and operational oceanography. Oceanography and Stewardship of marine ecosystems 527–560

Masria A, Negm A, Iskander M, Saavedra O (2014) Coastal zone issues: a case study (Egypt). Procedia Eng 70:1102–1111. https://doi.org/10.1016/j.proeng.2014.02.122

Mendez S, Ferrari G (2002) Harmful algal blooms in Uruguay: anteced-ents, ongoing projects and review of results. In Sar EA, Ferrario ME, Regueira B (Eds) Floraciones algales nocivas en el Cono Sur Americano, (In Spanish), pp 271–288

Mihnea PE (1985) Effect of pollution on phytoplankton species. Rapp Comm Int Mer Medit 29(9):85–88. https://publons.com/journal/115376/rapp-comm-int-mer-medit/

Mikhail SK (1997) Ecological Studies of the Phytoplankton in Mex Bay. PhD Thesis. Alexandria University, Egypt pp 266

Mikhail SK (2008) Dynamics of estuarine phytoplankton assemblages in Mex Bay, Alexandria (Egypt): Influence of salinity gradients. EJABF 12(4):231–251. https://doi.org/10.21608/ ejabf.2008.2014

Mingazzini M, Rinaldi A, Montanari G (1992) Multi-level nutrient enrich-ment bioassays on Northern Adriatic coastal waters. In Vollenweider RA, Marchetti R, Viviani R (Eds) Marine Coastal Eutrophication 115–131

Mizuno T (1990) Illustrations of the freshwater plankton of Japan. 9th Printing. Hoikusch Publishing Co. LT, Japan, p 353

Mohamed ZA (2016) Breakthrough of Oscillatoria limnetica and microcystin toxins into drinking water treatment plants–examples from the Nile River, Egypt. Water SA 42(1):161–165. https://doi.org/10.4314/wsa.v42i1.16

Moura AN, Araújo-Tavares NK, Amorim CA (2018) Cyanobacterial blooms in freshwater bodies from a semiarid region, Northeast Brazil: a Review. J Limnol 77(2):179–188. https://doi.org/10.4081/jlimnol.2017.1646

Murphy JA, Riley JP (1962) A modified single solution method for the determination of phosphorus in natural waters. Anal
Chim Acta 27:31–36. https://doi.org/10.1016/s0003-2670(00)88444-5
Nessim RB (1994) Environmental characteristics of Mex Bay. In First Arab Conference on Marine Environment Protection, Alexandria, 5–7 February 1994 p 221–243
Nessim RB, Bassiouny AR, Zaki HR, Moawad MN, Kandeel KM (2010) Environmental studies at El-Mex region (Alexandria-Egypt) during 2007–2008. World Appl Sci J 9(7):779–787
Ngatia L, Grace III, JM, Moriasi D, Taylor R (2019) Nitrogen and phosphorus eutrophication in marine ecosystems. In Monitoring of Marine Pollution. Fouzia, H. (Ed.) Intech Open
Okbah MA, Ibrahim AMA, Gamal MNM (2013) Environmental monitoring of linear alkylbenzene sulfonates and physicochemical characteristics of seawater in El-Mex Bay (Alexandria, Egypt). Environ Monit Assess 185(4):3103–3115. https://doi.org/10.1007/s10661-012-2776-9
Okbah MA, Masoud MS, El Zokm GM, Abd El-Salam AA (2017) Study of nutrient salts, chlorophyll-a and physicochemical condition in El-Mex Bay Water, Alexandria, Egypt. AJBES 3(4):54–64. https://doi.org/10.11648/j.ajbes.20170304.13
Orsini L, Sarno D, Procaccini G, Poletti R, Dahlmann J et al (2002) Toxic Pseudo-nitzschia multispirita (Bacillariophyceae) from the Gulf of Naples: morphology, toxin analysis and phylogenetic relationship with other Pseudo-nitzschia species. Eur J Phycol 37(2):247–257. https://doi.org/10.1080/0967229502003608
Palmer CM (1969) A composite rating of algae tolerating organic pollution. J Phycol 5(1):78–82. https://doi.org/10.1111/j.1529-8817.1969.tb02581.x
Peragallo M, Peragallo MMH (1897–1908) Diatomées de France etdes Districts Maritimes Voisins, 236 pp
Pucher-Petković T, Marasović I (1980) Développment des populations phytoplanctoniques caractéristiques pour un milieu eutrophisé (Baie de Kastela, Adriatique centrale). Acta Adriat 27:31–36. https://doi.org/10.1016/s0003-2670(00)88444-5
Rajesh KM, Gowda G, Menon M, Nazareth AP (2001) Distribution of sediment chlorophyll-a and phaeo-pigments in the brackish water ponds along the Nethravathi estuary, India. Indian J Fish Sci 48(2):145–149
Revelante N, Gilmartin M (1985) Possible phytoplankton species as indicators of eutrophication in the northern Adriatic Sea. Rapp Comm Int Mer Méditerr 29(9):89–91
Riley JP, Chester R (1971) Introduction to marine chemistry. Academic press, New York, p 465
Roubeix V, Lancelot C (2008) Effect of salinity on growth, cell size and silicification of an euryhaline freshwater diatom: Cyclotella meneghiniana Kütz. Transitional Waters Bull 2(1):31–38. https://doi.org/10.1128/i20152292Xv2n1p31
Said MA, El-Deek MS, Mahmoud TH, Shridah MMA (1991) Physicochemical characteristics of different water types of El-Mex Bay, Alexandria Egypt. Bull Natl Inst Ocean Fish ARE 17(1):103–116
Said MA, El-Deek MS, Mahmoud TH, Shridah MA (1994) Effect of pollution on the hydrochemical characteristics of different water types in El-Max Bay area, west of Alexandria, Egypt. Acta Adriat 34(1):9–19
Saito Y, Atobe S (1970) Phytosociological study of intertidal marine alge I. Usujiri Benten-Jima. Hokkaido. Bull Fac Fish Hokkaido Univ 21(2):37–69
Samaan AA (1974) Primary production in Lake Edku. Bull Nat Inst Oceanogr Fish A.R. Egypt 4:261–317
Scanlan CM, Foden J, Wells E, Best MA (2007) The monitoring of opportunistic macrogalgal blooms for the water framework directive. Mar Pollut Bull 55(1–6):162–171. https://doi.org/10.1016/j.marpolbul.2006.09.017
Shams El Din NG, Abo El Khair EM, Dorgham MM (2014) Phytoplankton community in the Egyptian Mediterranean coastal waters. Indian J Geo-Mar Sci 43(10):1981–1988
Shannon GE, Weaver W (1963) The mathematical theory of communication. University of Illinois Press, Urbana, p 117
Shaw GR, Moore DP, Garnett C (2003) Eutrophication and algal blooms. Environ Ecol Chem 2:1–21
Shobier AH, Abdel Ghani SA, Shreahda MA (2011) Distribution of total mercury in sediments of four semi-enclosed basins along the Mediterranean coast of Alexandria, Egypt. J Aquat Res 37(1):1–11
Shreahda MA, Masoud MS, Khattab ARM, El Zokm GM (2014) Impacts of different drains on the seawater quality of El-Mex bay (Alexandria, Egypt). J Ecol Nat Environ 6(8):287–303
Smetaacek V, Zingone A (2013) Green and golden seaweed tides on the rise. Nature 504(7478):84–88. https://doi.org/10.1038/nature12860
Smith VH, Joye SB, Howarth RW (2006) Eutrophication of freshwater and marine ecosystems. Limnol Oceanogr 51(1):351–355. https://doi.org/10.4319/lo.2006.51.1_part_2.0351
Soliman AM, Gharib SM (1998) Water characteristics, phytoplankton and zooplankton population of El-Mex bay region. Bull Fac Sci Alexandria Univ 38(1–2):45–66
Sournia A (1986) Atlas of marine phytoplankton. Introduction. Cyanophycées, Dicotychophycées, Dinophycées and Raphidio- phycées. Vol 1. Paris: Edition du CNRS, pp 216 (In French)
Sournia A (1968) Planktonic diatoms of the Mozambique Channel and Mauritius. Memoire Orstom, Paris, p 120 (In French)
Stachowitsch M, Avcin A (1988) Eutrophication-induced modifications of benthic Communities UNESCO/UNEP Workshop on Eutrophication in the Mediterranean: Receiving capacity and long term effects. UNESCO Re Mar Sci 49:67–80
Stirm J (1988) Eutrophication in the Mediterranean Sea: receiving capacity and monitoring of long-term effects. UNESCO, Technical reports series. Athens: (No 21), pp 195
Strickland JDH, Parsons TR (1968) A practical hand-book of seawater analysis. Bull Fish Res Board Canada 167:311. https://doi.org/10.1086/406210
Toming K, Jaanus A (2007) Selecting potential summer phytoplankton eutrophication indicator species for the northern Baltic Sea. Proceedings of the Estonian Academy of Science. Sci Biol Ecol 65(4):297–311
Tonk L, Visser PM, Christiansen G, Dittmann E, Snelder EO et al (2005) The microcystin composition of the cyanobacterium Planktothrix agardhii changes toward a more toxic variant with increasing light intensity. Appl Environ Microbiol 71(9):5177–5181. https://doi.org/10.1128/aem.71.9.5177-5181.2005
Turner AD, Dhanji-Rapkova M, O’Neill A, Coates L, Lewis A et al (2018) Analysis of microcystins in cyanobacterial blooms from freshwater bodies in England. Toxins 10(1):39. https://doi.org/10.3390/toxins10010039
Utermöhl H (1958) To perfect the quantitative phytoplankton method. Internationale Vereinigung für theoretische und angewandte Limnologie: Mitteilungen 9(1): 1–38. (In German)
Valderrama JC (1981) The simultaneous analysis of total nitrogen and total phosphorus in natural waters. Mar Chem 10(2):109–122. https://doi.org/10.1016/0304-4203(81)90027-x
Vucak ZAS, Sürn J (1982) Basic physical, chemical and biological data reports, RV. ‘A. Mohorovicic’Adriatic cruises 1974–76. Hydrogr Inst Yugoslav Navy, Split, pp 175
Wan AH, Wilkes RJ, Heesch S, Bermejo R, Johnson MP et al (2017) Assessment and characterisation of Ireland’s green tides (Ulva species). PLoS One 12(1). https://doi.org/10.1371/journal.pone.0169049

Wang J, Wu Y, Qin Y, Li Y (2008) The threat of potential alien species in the East China Sea. Dalian, P.R. China. 2008.10.28

Wassmann P, Olli K (eds) (2005) Drainage basin nutrient inputs and eutrophication: an integrated approach. University of Tromsø, Norway and Tartu University, Estonia, p 325

Whilm JL, Doris TC (1966) Species diversity of benthic invertebrates in a stream receiving domestic and oil refinery effluents. Ann Midl Nat 76(2):427–449. https://doi.org/10.2307/2423096

Wilson MA, Carpenter SR (1999) Economic valuation of freshwater ecosystem services in the United States: 1971–1997. Ecol Appl 9:772–783

Wong SC, Harah ZM, Sidik BJ, Arshad A (2012) Comparison of seaweed communities of the two rocky shores in Sarawak, Malaysia. Coast Mar Sci 35(1):78–84

Zaghloul FA (1994) Phytoplankton dynamics in the Western Harbour of Alexandria. Egypt. Bull Nat Inst Oceanogr Fish A. R. Egypt 20(2):107–117

Zakaria HY, Radwan AA, Said MA (2007) Influence of salinity variations on zooplankton community in El-Mex Bay, Alexandria, Egypt. Egypt J Aquat Res 33:52–67

Zalat A, El-Sheekh M, Ghandour I, Basaham AS (2021) Diatom assemblages from surface sediments of two coastal lagoons, central part of the Red Sea, Saudi Arabia and their associated environmental variables. Thalassas: An Int J Mar Sci 37:179–203

Zanchett G, Oliveira-Filho EC (2013) Cyanobacteria and cyanotoxins: from impacts on aquatic ecosystems and human health to anticarcinogenic effects. Toxins 5(10):1896–1917. https://doi.org/10.3390/toxins5101896