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Bioassessment of water quality of surface waters using diatom metrics

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Abstract: Bioassessment of surface waters is one of the most important approaches to predict the deterioration of ecosystems and achieve environmental sustainability according to the application of the European Water Framework Directive. The present review emphasizes the importance of the bioassessment of freshwater quality especially running waters based on diatom metrics. Nutrient enrichment and hydromorphological alternation driven by human activities are the main factors for the ecological compromise of freshwater ecosystems. Currently, the bioassessment of the ecological condition of inland water bodies is adopted worldwide. Bioassessment is complementary to physicochemical and hydromorphological data for evaluating the ecological conditions of rivers; however, measuring all the physical and chemical changes is expensive and impractical. Therefore, monitoring biota helps to determine the changes occurring in ecosystems. Thus, diatoms are used as bioindicators to assess environmental conditions of the ecosystems, but their use requires great taxonomic knowledge, otherwise, the results will be biased. Many diatom indices have been developed based on the trophic weight and indicator values of diatoms in different ecoregions in the last decades. This review highlights the importance and advantages of using diatom metrics in the bioassessment of the ecological status of surface waters in the different ecoregions, especially running water. To analyze the complex response of diatom communities to environmental gradients and assess the quality of the ecosystem, multivariate statistical approaches are needed.
The challenge here is how to define criteria for classes of water bodies in a biologically meaningful way. For this reason, biological condition gradient is suggested as an appropriate and effective approach to develop trophic criteria based on the relationships between nutrient concentrations and biological indicators of ecological conditions.

Key words: Biological condition gradient, diatom metrics, ecoregion, water quality evaluation

1. Introduction

Freshwater is a vital resource for biosphere life and the functions of ecosystems that also support human well-being (Heinze et al., 2020). Availability of freshwater not only affects the distribution of biota on the earth, but also affects expanding human populations, and human activities like irrigation, industry, drinking, transportations, recreation, and fishery.

Streams and rivers are directly related to settlements and land-uses and so, they are intimately affected by stressors from human activities. For that reason, healthy downstream streams and rivers need healthy headwater streams (Tilman et al., 2014) because they can store and transform nutrients. Thereby, water further downstream would have increased amounts of nitrogen and phosphorus that cause eutrophication in aquatic ecosystems. Accordingly, biological and chemical monitoring of water resources gives important knowledge about the environmental conditions for the provisioning of ecosystem services (Bullock et al., 2011; Ccanccapa et al., 2016). In addition, monitoring supports the objective decisions necessary to understand the strength of policies created to improve the current and future water quality.

Expansion of human activities (e.g., urbanization, wastewater disposal, agricultural land-uses, modifications, combustion of fossil fuels), increasing human populations and changes in global climate have extensively altered the freshwater ecosystems by the modifications impacting the physical, chemical, and biological features (Çelekli and Lekesiz, 2020). These anthropogenic factors lead to the deterioration of freshwater resources that create many problems in the allocation of equal and sustainable water to those who benefit from resources (Best, 2019; Freitas et al., 2020).

Although the availability of freshwater is limited on earth, existing resources are rapidly polluted by human activities (Figure 1). Functions of ecosystems depend on healthy environments and the ecological balance, which

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Figure 1. Effects of human activities on the states and functions of aquatic ecosystems.

strongly support the biota on the biosphere. Therefore, high water quality is one of the most important factors affecting biota throughout the food web to prevent health problems of living forms (e.g., humans). All these factors indicate the equal importance of the protection, renewable, and sustainable use of water (Torres-Franco et al., 2019). Thereby, natural freshwater resources should be regularly monitored to protect and enhance the qualities of disturbed water ecosystems and to achieve a good ecological status (EU WFD, 2000). Investigation and decreasing/preventing the kind of pressures on natural resources/ecosystems are the most important elements in the long-term realization of sustainable use.

Nutrient enrichment and hydromorphological alteration especially from human activities are major factors for the ecological deterioration of freshwater ecosystems (Gao et al., 2020; Zohary et al., 2020). Adverse effects of nutrient enrichment on water systems have been widely reported, but knowledge on hydromorphology is still not sufficient. Although hydromorphology has multiple definitions (Vogel, 2011), the word morphology was effectively used after Water Framework Directive (WFD) as the morphological quality elements supporting the biological quality components. The concept of hydromorphology has been formed by the combination of geomorphology, hydrology, and ecology (Vogel, 2011). Hydromorphology is expressed as a subfield of hydrological engineering that addresses the problems related to the structure of hydrological systems, shape, boundaries, evolution, and dynamic morphology of hydrological systems (EU WFD, 2000; Vogel, 2011). For that reason, hydromorphology is showed as one of the main components of ecosystem characteristics (Kelly et al., 2019). Hence, hydromorphological monitoring is carried out not only to identify hydrological and physical interventions impacting aquatic ecosystems but also to learn about the biological conditions of water for supporting the assessment of the ecological status of aquatic ecosystems (Stevenson and Pan, 1999; Stefanidis et al., 2019).

Nowadays, the bioassessment of the ecological conditions of inland water bodies is adopted worldwide (EU WFD, 2000; Charles et al., 2021). Because the results of bioassessment are complementary to physicochemical and hydromorphological data for evaluating the ecological conditions of rivers. The present review aimed to qualitatively highlight the importance of bioassessment of freshwater quality especially for running waters (e.g., creeks, streams, and rivers) based on diatom metrics, since the biological evaluation approach is becoming more important to interpret the deterioration of aquatic ecosystems. Also, the development of bioindicator, typology, ecoregion, and diatom indices studies in freshwater monitoring will be explained in this review. The absence of diatom index studies using the biological condition gradient (BCG) in the literature indicates that there is a gap in the bioassessment of ecological conditions of lotic ecosystems. Therefore, it was emphasized that the results of biological, chemical, and hydromorphological evaluations should be supported with the BCG approach to be more accurate and applicable. The BCG is a comprehensive, descriptive, scientific, and ecosystem-based framework that describes a gradient in resource conditions including biological, physical, and chemical variables to standardize biological assessments of freshwater streams (Hausmann et al., 2016; Charles et al., 2019; Ruaro et al., 2020).

2. Data sources

Google Scholar, Scopus, SpringerLink, Web of Science, and Mendeley databases were searched to reach appropriate publications to support the goal of this study, using "assessment", "bioindicator", "diatom indices", "ecoregion", "biological condition gradient", "WFD", and combinations of them. In this review, the main criteria is to get the results of the diatoms-based publications since diatoms are the main biological group whose importance is being emphasized. To reach such kinds of publications, the keywords such as bioassessment, bioindicator diatom species, ecoregion concept, diatom trophic indices, and water quality based on diatoms were researched on the aforementioned databases. In addition to these key words, ecosystem types were searched for lotic, lentic, basins, and surface waters. After this extensive research more than tens of thousands of publications (e.g., research articles, short communications, reviews, etc.) were found. And then, we narrowed down the search to diatoms for highlighting the importance of the bioassessment based on the diatom metrics in the present review.

3. Bioassessment of surface waters

The biological assessment of surface water quality has been a more important issue to evaluate the deterioration of aquatic ecosystems and to accomplish environmental sustainability since the implementation of the EU Water Framework Directive. Biomonitoring programs based on diatom assemblages can be an interesting and promising approach for the evaluation of aquatic ecosystems. The present study emphasized that the results of bioassessment are complementary to physicochemical and hydromorphological data for evaluating the ecological conditions of rivers.

Physicochemical measurements are carried out to determine water quality, water efficiency, environmental conditions, and sustainability of surface waters at the time of measurements (EC, 2009; Toudjani et al., 2017; Dalu et al., 2020; Charles et al., 2021). General physicochemical parameters such as nutrients (TP, TN, PO_4, NO_3, etc.), heavy metals, xenobiotic, dissolved oxygen, electrical conductivity, and pH should be measured during water quality assessment studies (EC, 2009; Charles et al., 2021). According to the WFD, European member states are obliged to apply these parameters, which are also recommended to be used to determine water quality. However, WFD member states have the right to select some of these parameters (Quevauviller, 2006).

Chemical assessment of surface waters is one of the relevant ways, but this provides limited information about the ecological status of aquatic ecosystems. Unless chemical assessment is done regularly, it does not reflect the changes that may occur over time, as it will only give instantaneous values. For this reason, it may not provide complete information on the real conditions of aquatic ecosystems (Bere and Tundisi, 2011; Brack et al., 2019; Çelekli et al., 2019a; Charles et al., 2021). Deterioration of aquatic ecosystems is not only based on factors such as pesticides, xenobiotic, salinity, turbidity, and smell but also there are hundreds of other chemical variables (Merga et al., 2020). Measuring all of them can be very expensive and impractical. Any fluctuations in abiotic conditions of aquatic ecosystems will cause changes in biota. Thereby, monitoring biota helps to determine the changes occurring in the ecosystems. Therefore, an important endeavor has been put forward to assess water quality by using the combination of biological and physicochemical variables in recent decades (Rott et al. 2003; Lobo et al. 2015, 2016; Çelekli et al. 2018; Çelekli and Arslanargun, 2019).

Biological monitoring indicates the cumulative effects of all environmental parameters on the ecosystems (Lobo et al., 2004, 2016, 2019; Çelekli and Kapı, 2019; Ballesteros et al., 2020; Mbao et al., 2020; Pajunen et al., 2020). The water quality and long-term scale changes observed by using the biological assessment should be supported by physical and chemical approaches (Çelekli and Kapı, 2019; Lobo et al., 2019; Mbao et al., 2020; Pajunen et al., 2020). On the other hand, there are some challenges in biological monitoring, and they should not be ignored. These challenges are i) the bioassessment requires great taxonomic expertise and it is difficult to find young researchers who become experts in taxonomy with the requirement of many years, ii) developing reference conditions for sectional studies requires intensive effort and good design and iii) choosing of the best time for sampling that may vary from a region to another

Water quality assessment using biological indicators began in the twentieth century (Bauernfeind and Moog, 2000). Therefore, assessing pollution in ecosystems using biological quality components is not a new approach. The concept of expressing aquatic conditions with a biological approach began with the work of Kolkwitz and Marson in Germany in 1909 (Kolkwitz and Marson, 1909) that was developed as the saprobic system. The saprobic system is based on water quality and the degree of pollution due to the responses of benthic organisms such as *Drunella* (Ephemerellidae), Plecoptera, and *Rhyacophilla* (Rhyacophilidae) in oligosaprobic streams; *Hydropsyche kozhantschikovi- Uracanthella rufa - Epeorus latifolium* in mesosaprobic; and *Chironomus yoshimatsui* group in-mesosaprobic streams and Tubificidae in polysaprobic streams (Bae et al., 2005) and also in different ecosystems with different amounts of organic pollution (Bauernfeind and Moog, 2000). The saprobic system has been revised, developed, and expressed in mathematical systems, which is still being used because scientists have started to use numerical data to assess the impacts of pollutions on organisms since the beginning of the 20th century (Zelinka and Marvan, 1961; Sládeček, 1986).

3.1. Legislation of clean water act

Many countries around the world have held meetings to provide holistic environmental management, pollution control, and environmental problems due to the pressures in aquatic ecosystems. In 1948, the Federal Water Pollution Control Law (FWPCA) was adopted in the United States to control the discharge of pollutants into U.S. waters and to set the quality standards required for surface waters. The FWPCA as revised in 1972 aims to protect the physical, chemical, and biological integrity of surface waters. From 1972 to 1977, it was largely regulated and expanded by a law entitled Federal Water Pollution Control Act Amendments (Chakraborti and McConnell, 2012). Since then, the clean water act (CWA) has been a common name for laws enacted to protect the integrity of the country's waters. Its laws and regulations are governed primarily by the U.S. Environmental Protection Agency (EPA) in coordination with state governments (USEPA, 2016). Since 1990, more than twenty directives have been presented for the ecological quality and the sustainable use of streams around the world (Hering et al., 2010; Freitas et al., 2020).

3.2. European Water Framework Directive

The European Water Framework Directive (WFD) was announced on 23 October 2000 (EU WFD, 2000). The WFD offers a framework for the protection of coastal waters, transit waters, and continental surface waters (EU WFD, 2000). WFD's main goal is to prevent, protect, and improve the ecological state of aquatic ecosystems. Freitas et al. (2020) reported that the WFD aims the rational use of water resources, conservation, and improvement of the quality of aquatic systems (surface, estuarine, coastal, and underground waters) to reach good ecological status until the year 2021. This announcement of WFD also asked its members to guarantee the gradual reduction of pollutants on the ecosystems for the longer term protection of aquatic environments. The WFD provides a political basis for all public and local authorities to ensure that all surface waters reach a good ecological and chemical level based on the biological, chemical, and hydromorphological evaluations (Andersen et al., 2004; Charles et al., 2021). The WFD implemented three monitoring programs: operational, surveillance, and investigative to evaluate the overall surface water status in the river basins for achieving different environmental objectives. About 110,000 stations of water bodies in the EU have been monitored and the largest number of them cover rivers (about 80%) with 67,691 sites of operational monitoring and 19,637 sites of surveillance monitoring (EC, 2009). The main goals of these bioassessment monitoring are to determine the ecological status of water resources and to understand how natural and anthropogenic factors affect ecosystems. The WFD classified surface waters according to their ecological quality as following high, good, medium, poor, and bad (EU WFD, 2000). Surface waters are mainly categorized into lakes, rivers, transit waters, coastal waters, artificial bodies, and largely modified water bodies. The typological criteria are altitude, geology, slope, and precipitation (EU WFD, 2000) for the classification of lotic and lentic ecosystems. The flow regime in the streams is evaluated according to the seasonal and continuous flow status. Altitude (0–800 m, 800–1600 m, and >1600 m), slope (<2% and more than 2%), geology (high and low mineralization), drainage area (wet and dry regions), and precipitation (<400 mm and more than 400 mm) are the important typological criteria for running waters. Along with the altitude and geology, surface area (50 ha, 50<–<500 ha, and >500 ha) and depth (up to 5 m and >5 m) are also used as the typological criteria for lentic ecosystems.

Ecosystems linked to typologically classified water bodies where stress factors such as agriculture, industry, and human influences are seen at the lowest level and/or undisturbed, are considered as reference areas (Andersen et al., 2004). Then, type-specific reference areas are determined for the ecological assessment of each water body. Besides, the number of reference areas should be worth reflecting the structure of water bodies to which it depends. After the determination of the reference areas, biological, hydromorphological and physicochemical quality variables of sampling stations and their typologically related reference areas have to be measured for the bioassessment of ecosystems (Moldoveanu et al., 2017; Charles et al., 2021). And then the ecological quality ratio (EQR) of surface waters can be calculated based on the comparison of the measured metric (observed) in the water bodies with expected metric value in the specific reference area (e.g., EC, 2009; Hering et al., 2010; Lazaridou et al., 2018; Çelekli et al., 2019b). Thus, EQR outcomes as observed/expected metric values. After the normalization analysis, EQR is expressed by numerical data in the range of 0–1. While the high ecological status is represented with values close to 1, the bad environmental condition is showed by the values equal or close to zero (0) (EC, 2009). Boundaries of ecological status given in Annex V of the WFD are generally represented in the range of 1.0–0.8 for high, 0.8–0.6 for good, 0.6–0.4 for moderate and 0.4–0.2 for poor, and as bad at <0.2 (Figure 2). These class boundaries can show changes in different ecological regions.

There is the minimal human impact on the hydromorphological and physicochemical parameters in the high-ecological status waters (EC, 2009; Charles et al., 2021). The anthropogenic effects are at low levels for the good ecological status of aquatic ecosystems (EC, 2009). Moderate ecological quality waters are subject to moderate deterioration depending on human activities. Therefore, moderate ecological quality waters cannot be used in industries like food and textiles. However, it can be used in other industrial areas after the proper water treatments. Human activities have strong effects on the poor ecological quality waters (EC, 2009; Toudjani et al., 2017). These water bodies are often used in agricultural irrigation, and it is not suitable for drinking water use even if disinfected. Bad ecological status waters are not suggested for drinking or agricultural purposes and industrial process (EC, 2009). Mentioned water class boundaries can also be selected in nature. Some views of streams with different ecological statuses in different ecoregions are given in Figure 3. The least anthropogenic effects, no settlement, and land-use around streams (Figures 3a and 3b) in the Aras river basin have high ecological status based on the diatom metrics (Çelekli et al., 2019a). The stream in Figure 3c has a moderate ecological status, while the system in Figure 3d shows a good environmental condition in the Antalya river basin. The streams (Figures 3e and 3f) in the Ceyhan river basin under pressures of human activities have poor environmental conditions.

3.3. Ecoregion

The term ecoregion was used for the first time by Canadian forest researcher Loucks (1962). Ecological zones are well-known geospatial units for conservation planning developed to express models of ecological and environmental variables affecting the distribution of biodiversity characteristics at large scales (Omernik and Griffith, 2014). The term ecoregion was initially used

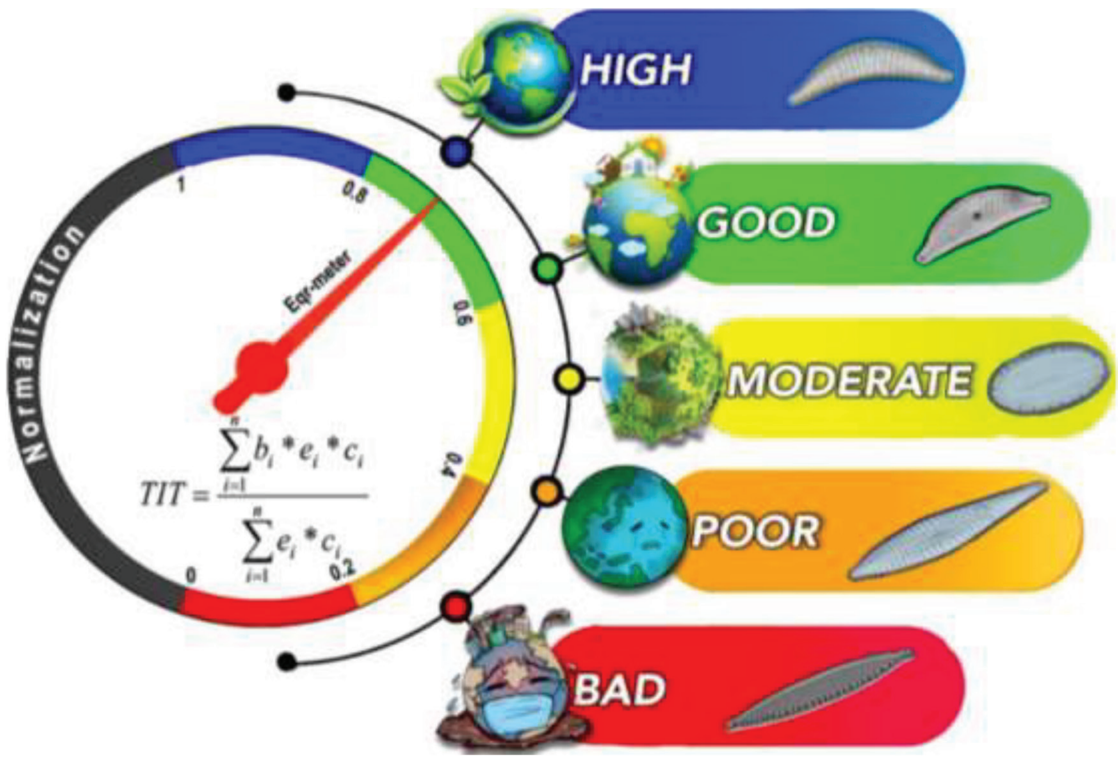

Figure 2. Boundaries of ecological status of surface water bodies.

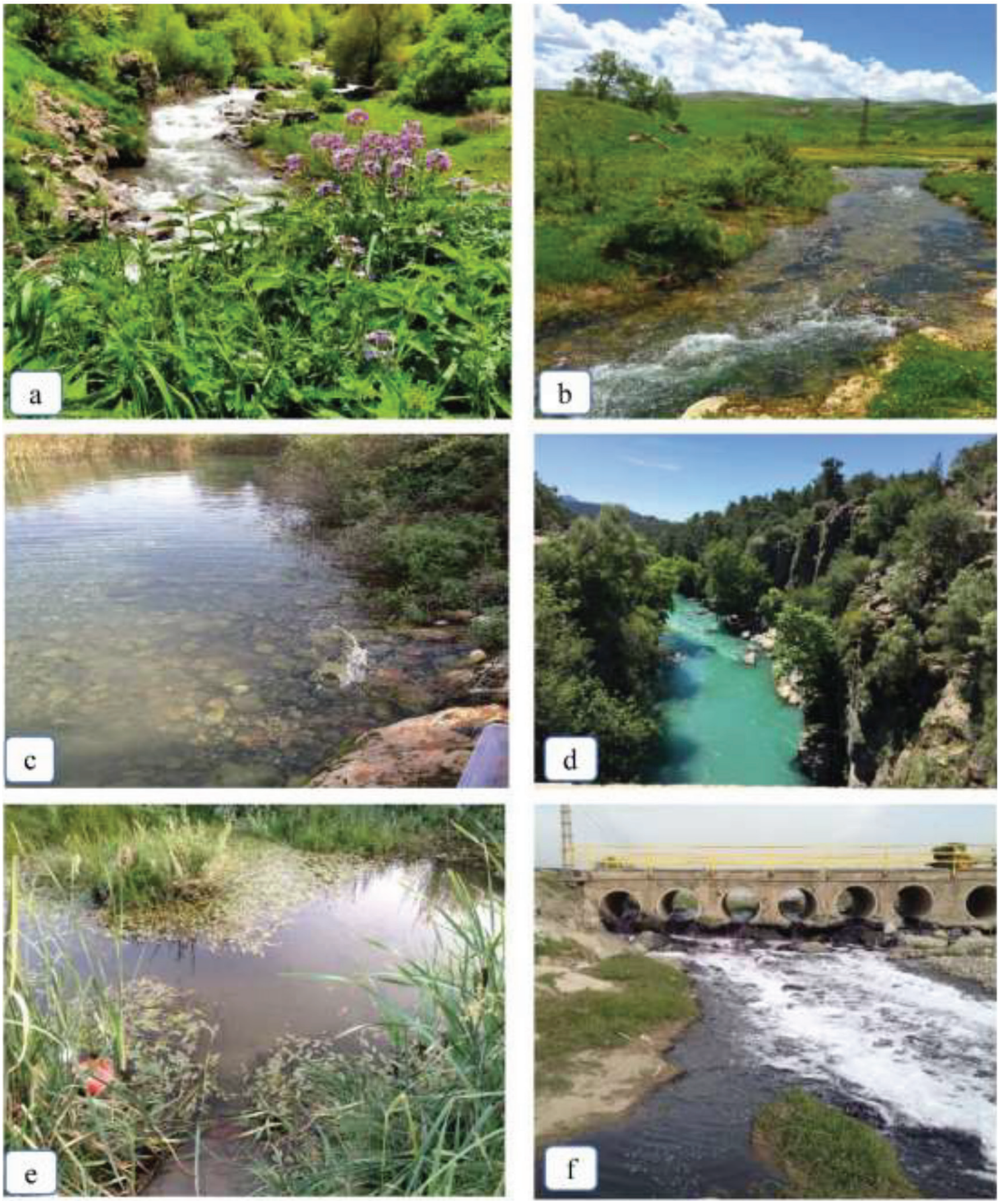

Figure 3. Different ecological status of lotic ecosystems from Aras (a and b), Antalya (c and d), and Ceyhan (e and f) basins in Turkey.

only to describe terrestrial ecosystems, while Omernik (1987) used the term for the grouping of similar aquatic ecosystems. In freshwaters, approaches for delineating ecoregions are based on clustering sites based on taxa present (Abell et al., 2008; Letten et al., 2018; Smith et al., 2020) or similarity in surrounding environmental variables (Omernik, 1987). The use of environmental data outside the aquatic ecosystem to determine the quality of wetlands is based on the idea that ecosystems are regularly and systematically affected by these data (Omernik, 1987; Kong et al., 2013; Poulíčková and Manoylov, 2019). During these definitions, the most used data are soil structure, land-uses, climate, altitude, geology, and hydrology data (Omernik, 1987; Higgins et al., 2005).

3.4. Diatom assemblages as bioindicator

Occurrences/successions of biological quality organisms like diatoms in the reference areas have become a necessity to compare other ecosystems in the bioassessment approach

(Rott et al., 1999; Dell'Uomo, 2004; Kelly et al., 2008). Reference values are required to classify the ecological quality status of the limited water resources. Another requirement is the assessment of the physicochemical and hydromorphological properties of the surface waters. This is because it is a complementary and supporting tool to the bioassessment of water bodies (EC, 2009; Hering et al., 2010; Toudjani et al., 2017).

Industrial changes around the world have also increased the diversity of pollution in aquatic ecosystems (Guittonny-Philippe et al., 2014). Increasing pollution causes a lot of waste; especially heavy metals blending into aquatic environments affect the living creatures in aquatic ecosystems. These undesirable compounds not only affect biota but also change the function of ecosystems. Besides, these chemical compounds can become more harmful by interacting with each other and disrupt the quality of the aquatic ecosystem (Guittonny-Philippe et al., 2014).

The number of studies dealing with the quality of aquatic bodies using bioindicators has started to increase. However, the important point in the bioassessment study is the difficulty of evaluating each pollutant and the choosing of correct bioindicator organisms. Hence, many pollutants can be evaluated when water quality monitoring is done with biological quality groups like diatoms and benthic macroinvertebrates (Van Dam et al., 1994; Salomoni et al., 2011; Lobo et al., 2019; Çelekli and Lekesiz, 2020).

The intense effects of deteriorating water quality on living things in aquatic ecosystems have led to the development of some concepts based on the bioindication system. Biological quality organisms are used as keys to assess past, future, or current environmental conditions of ecosystems. Bioindicators in the aquatic ecosystem give crucial responses to pollution gradient, which can provide valuable information about health and the overall ecological status of environments (Rott et al., 1999; Kelly et al., 2008; Birk et al., 2012; Toudjani et al., 2017; Çelekli et al., 2019b). Concerning that, tolerances and sensitivities of bioindicator species have been used to investigate and assess the potential effects of environmental pollutants on living things (Rott et al., 1999; Kelly et al., 2008; Çelekli et al., 2019b; Cozea et al., 2020). The advantage of using bioindicators is the integration of these organisms in the evaluated habitats (Lobo et al., 2016; Cozea et al., 2020). Also, they show universal behavior in different ecosystems, which is important for comparing pollution in different continents or ecosystems (Rott et al., 1999; Smol and Stoermer, 2010; Lange-Bertalot et al., 2017; Morales et al., 2020). Taxonomy and classification of bioindicator species must be carried out by specialist researchers, as a wrong classification makes all results wrong. Additionally, bioindicators should be sensitive to environmental changes and have their unique optimum tolerance levels to certain nutrients (Lobo et al., 2016; Salmaso et al., 2019).

Phytobenthos as a primary producer are found in benthic parts of aquatic ecosystems (Smol and Stoermer, 2010) and mostly consist of diatoms (Van Dam et al., 1994; Toudjani et al., 2017), which play an extremely important role in biogeochemical cycles. Due to their photosynthesis abilities, they increase and contribute to the dissolved oxygen used by living creatures in the aquatic ecosystem. Since they absorb harmful substances in ecosystems, they also act as filters for cleaning polluted water (Smol and Stoermer, 2010). Diatoms are also known as siliceous algae (Lowe, 1974) and therefore they are easily fossilized. Diatoms represent a diverse group with as many as 100,000 species that arose in the early Mesozoic as shown by both fossil and molecular data (Medlin, 2016).

European Water Framework Directive (WFD) points out that biological quality elements [e.g., phytobenthos (especially benthic diatoms), benthic macroinvertebrates, phytoplankton, macrophytes, and fish] can be employed as eco-indicators for the assessment of surface waters (EC, 2009). Among them, diatoms are commonly used in the bioassessments of the ecological status of lotic ecosystems due to their short life cycles and rapid response to different stressors in any season with a broad spectrum from very good to poor environmental conditions (Delgado and Pardo, 2014; Lobo et al., 2015, 2016, 2019; Çelekli et al., 2019b). Their taxonomy has been well documented since the important developments have been found concerning diatom identification in the last 10 years (e.g., Krammer, 2003; Guiry and Guiry, 2015; John, 2015; Levkov et al., 2016; Lange-Bertalot et al., 2017). Also, the taxonomy of diatoms can be demonstrated by the updated diatom databases such as EDDI (2012), Algaebase (Guiry and Guiry 2015), Diatoms of the United States (2016), and OMNIDIA (2017). Most recently, Ballesteros et al. (2020) studied genetic barcoding of epilithic diatom species as bioindicators to evaluate water quality.

Their sensitivity to the physicochemical (e.g., nutrients, electrical conductivity, salt, temperature, biological oxygen demand, etc.) changes in the different environments allow them to react very quickly to the spatial and temporal changes in environments (Descy and Coste, 1991; Rott et al., 1999; Dell'Uomo, 2004; Kelly et al., 2008; Lobo et al., 2015; Toudjani et al., 2017; Çelekli et al., 2019b; Huang et al. 2019; Park et al., 2020). They provide crucial information about the environment where they live because each taxon has different environmental optima for different pollutants of ecosystems (Stevenson and Pan, 1999; Kelly et al., 2008). Consequently, diatoms are very useful in biological monitoring studies (Martin and Reyes, 2012; Çelekli et al., 2019a). Knowing the indicator characteristics of diatom assemblages is critical for the robust inferences of the environmental conditions in monitoring programs and paleolimnological applications (Juggins and Birks, 2012).

All of them support that diatom assemblages as ecological indicators are widely used to estimate the ecological status of aquatic ecosystems in different ecoregions (Rott et al., 1999; Dell'Uomo, 2004; Smol and Stoermer, 2010; Lobo et al., 2016; Ruwer et al., 2018; Çelekli et al., 2019b; Pinheiro et al., 2020; Pham, 2020). Additionally, diatoms occur in almost all surface waters at all times of the year, which is the biggest advantage of the use of diatoms when determining the ecological quality of water bodies (Ács et al., 2004; Smol and Stoermer, 2010; Çelekli et al., 2018). Also, very strong correlations between stressors and diatoms confirm the highly accurate assessment of ecological conditions (Toudjani et al., 2017; Chen et al., 2019).

3.5. Diatom indices

Many studies have been carried out to determine the trophic weight and indicator levels of diatom assemblages in different ecoregions (e.g., Rott et al., 1999; Dell'Uomo, 2004; Kelly et al., 2008; Lobo et al. 2004, 2015; Çelekli et al., 2019b; Salinas-Camarillo et al., 2020). Responses of epilithic diatom assemblages to pollution gradient give crucial information to predict the health of lotic ecosystems (Lobo et al., 2010; Böhm et al., 2013; Heinrich et al., 2015; Castillejo et al., 2018). According to their occurrence/succession in different environmental gradients, diatom taxa are considered as pollution sensitive, intermediate pollution tolerant, and pollution tolerant species based on their trophic weights from various diatom indices (Table 1). Special tolerance levels of each diatom taxon to nutrients (e.g., phosphate, nitrogen, and other stressors) (Rott et al., 1999; Potapova and Charles, 2007; Munn et al., 2018; Dalu et al., 2020), allow scientists to develop diatom indices for monitoring and determining the quality of surface waters (see Table 2). Diatom-based indices offer a more stable approach than that of fishes and benthic macroinvertebrates due to the direct response of diatoms to stressors (Carlisle et al., 2008). Another difference of diatoms from other indicator organisms is the relatively low sampling costs. Besides, diatoms taken from the aquatic environment can be cheaply and easily stored for reexamining.

The study of Kolkwitz and Marson (1909) is accepted as pioneer research for index developing studies. Many indices have been developed based on the trophic weight and indicator values of the diatom species in different ecoregions in the last decades (e.g., Cemagref, 1982; Coste and Ayphassorho, 1991; Rott et al., 1999; Dell'Uomo, 2004; Lobo et al., 2004, 2015; Kelly et al., 2008; Benito et al., 2018; Çelekli et al., 2019b). To assess the ecological status of water bodies, DPI-(Descy's pollution index) in France (Descy, 1979), PSI-(pollution sensitivity index) in France (Cemagref, 1982), SI-(Sládeček's index) in Czechia (Sládeček, 1986), TI-(trophic index) in Austria (Rott et al., 1999), EPI-D-(eutrophication/pollution Index) in Italy (Dell'Uomo, 2004), TDI-(trophic diatom index) in England (Kelly et al., 2008), TWQI-(trophic water quality index) in Brazil (Lobo et al., 2015), TIT-(trophic index of Turkey) in Turkey (Çelekli et al., 2019b), and DEQI (diatom ecological quality index) in Mexico (Salinas-Camarillo et al., 2020) have been developed in the different ecoregion of world. Developed diatom indices are given in Table 2. Diatom indices are mostly based on the equation of Zelinka and Marvan (Zelinka and Marvan, 1961), which take into account the types of stream pollution (i.e. salinity, nutrients, pH, BOD, etc.) and weighted taxon sensitivity averages. The direct use of indices developed in different ecoregions for a specific country may produce erroneous results to assess the state of water quality (Tomas et al., 2017; Çelekli et al., 2018; Riato et al., 2018). Geographical variations among countries, differences in human population density, land uses extent (agricultural, industrial, urban), and climate can be shown as the obstacles for the direct use of these indices to accurately interpret the water quality (Soininen, 2007; Çelekli et al., 2018; Charles et al., 2021). Ecoregional variation (e.g., geology, climate, land-uses, and anthropogenic activities) in countries can constraint and regulate diatom composition and their abundance. The fluctuation of ecological preferences of diatom assemblages according to the temporal and spatial changes is a well-known phenomenon and so each index gives the most accurate and reliable result for the country where it has been developed.

Pollution tolerant and sensitive diatom assemblages accurately indicate the variation in environmental conditions under human disturbance with a loss of sensitive species or an increment of tolerant species (Davies and Jackson, 2006). Strong relationships between diatoms and stressors (e.g., nutrients especially soluble reactive phosphorus and TP electrical conductivity, salinity, acidity, etc.) are quantifiable in the different trophic gradients from reference sites to highly disturbed sites. Occurrences/successions of diatom assemblages in different environmental conditions are deciphered by using multivariate complex statistical analyses to determine their trophic weight and indicator values. The gathered information on state-specific metrics derived from species optima and stressor response model is used in various biological metrics to assess the ecological status of water bodies (e.g., Karr and Chu, 1998; Rott et al., 1999; Dell'Uomo, 2004; Kelly et al. 2008; Lobo et al., 2015; Çelekli et al., 2019b; Salinas-Camarillo et al., 2020). Charles et al. (2021) reported that the most commonly used metrics are associated with reactive phosphorus, total phosphorus, total nitrogen and nitrate-nitrogen. Besides, organic pollution is also explained by the term "general degradation" according to the biological oxygen demand and dissolved oxygen demand parameters. The

Table 1. Trophic weights of some diatom species according to their sensitivity and tolerance to pollution gradient. TI-trophic index (Rott et al., 1999), EPI-D-eutrophication/pollution index (Dell'Uomo, 2004), TDI-trophic diatom index (Kelly et al., 2008), TIT-trophic index of Turkey (Çelekli et al., 2019b), TWQI-trophic water quality index (Lobo et al., 2015), and RRDI-Richmond River diatom index (Oeding and Taffs, 2017).

Pollution sensitive species	Diatom index
Achnanthidium minutissimum	TI, TIT,EPI, TDIL, TWQI
Achnanthes microcephala	TI, TIT,EPI
Adlafia bryophila	TI, TIT,EPI
Cymbella affinis	TI, TIT,EPI, TDI
Cymbella excisa	TI, TIT,EPI, TDI
Cymbella microcephala	TI, EPI, TDI
Denticula kuetzingii	TIT, TI
Denticula tenuis	TI, TIT,EPI
Diatoma tenuis	TIT, TI
Epithemia turgida	TIT, EPI
Fragilaria tenera	TI, TIT,EPI
Hannaea arcus	TI, TIT,EPI
Navicula radiosa	TI, TIT,TDIL
Odontidium mesodon	TI, TIT,EPI, TDI
Tabellaria fenestrata	TI, EPI, TDI
Intermediate pollution tolerant species	Diatom index
Amphora inariensis	TI, TIT,
Aulacoseira italica	TIT, EPI
Bacillaria vulgaris	TIT, TI
Caloneis silicula	TI, TIT,EPI
Cocconeis lineata	TI, EPI, TDI
Cocconeis placentula	TI, TIT,EPI, TDI
Cymbella neocistula	TI, TIT,EPI
Gomphonema minutum	TI, TIT,EPI, TDI
Fragilaria construens	TI, TIT,EPI
Fragilaria dilata	TI, TIT,EPI
Fragilaria rumpens	TDIL, TWQI
Fragilaria capucina	TI, TIT,EPI
Gomphonema acuminatum	TI, EPI, TDI, RRDI
Gomphonema angustatum	TIT, EPI, TDI, TWQI, RRDI
Nitzschia fonticola	EPI, TDI
Pollution tolerant species	Diatom index
Gomphonema augur	TI, TIT, EPI
Gomphonema pseudoaugur	TDIL, TWQI
Navicula cincta	TI, TIT, EPI
Navicula cryptocephala	TI, TIT, EPI
Navicula menisculus	TI, TIT, EPI
Navicula recens	TI, EPI, TDI

Table 1. (Continued).

Nitzschia calida	TI, TIT, EPI, TDI
Nitzschia linearis	TI, TIT, EPI, TDIL, TWQI
Nitzschia palea	TI, TIT, EPI, TDI, TWQI, RRDI
Nitzschia frustulum	TI, TIT, EPI, TDIL
Planothidium lanceolatum	TI, TDI
Pinnularia viridis	TIT, EPI
Surirella brebissonii	TIT, EPI
Tryblionella calida	TI, TIT,EPI
Rhoicosphenia abbreviata	TI, TIT,TDIL

pollution sensitivity index (IPS) is a widely used diatom index incorporating nutrients and organic pollution, and Rott trophic index (TI) and trophic diatom index (TDI) are related to total phosphorus (TP) and soluble reactive phosphorus, respectively. The setting of standards for physicochemical factors shows variability in different countries such as acidity in Sweden, salinity and heavy metals in Belgium, organic matter in Italy, Poland, and Slovenia, TP in Austria, Estonia, and Turkey.

The bioassessments require the selection of the best metrics having a good response to the gradient of human impacts and so they are selected based on their attributions like followings: (i) metrics should have low variation among reference areas and should show significant differences between highly disturbed ecosystems and type-specific reference site, (ii) metrics should have a trophic gradient greater than zero, (iii) they should display variability among different trophic sites, etc. (Rott et al., 1999; Dell'Uomo, 2004; Kelly et al. 2008; Lobo et al., 2015; Çelekli et al., 2019b; Charles et al., 2021).

Developed different diatom indices over the world are given in Figure 4. In the first view, some diatom indices have been developed but they are not adequate when considering ecoregions in the world without indices. The gray-colored countries have not developed specific diatom indices to assess the ecological status of their surface waters, and some of them are used developed and/or modified indices. However, these countries have different ecoregions including geology, climate, vegetation, wildlife, hydrology, and human activities (Omernik, 1987; Çelekli and Kapı, 2019; Espinosa et al., 2020), which strongly affect the environmental factors on the trophic weight and indicator values of diatom taxa (Lobo et al., 2004, 2015; Çelekli et al., 2019b; Salinas-Camarillo et al., 2020). Thereby, using foreign diatom indices can lead to erroneous interpretation of water quality. Hence, ecoregional specific diatom metrics are needed to accurately determine the surface water quality.

Research dealing with bioindicators has gained momentum to understand the responses of species to stressors and such kinds of studies indicate an uptrend. Many bioassessment studies dealing with diatom metrics have been carried out in Europe, Asia, and America (e.g., Lavoie et al., 2009; Wachnicka et al., 2011; Bere, 2016; Vilmi et al., 2016; Xue et al., 2019; Szczepocka et al., 2019; Tapolczai et al., 2019; Edwards et al., 2020), while there are a few studies in Turkey, e.g., in the West Mediterranean basin (Çelekli and Lekesiz, 2020), Aras basin (Çelekli et al., 2019a), Sakarya basin (Çetin and Demir, 2019) and the southeast of Anatolia (Çelekli and Arslanargun, 2019; Çelekli and Bilgi, 2019; Çelekli and Kapı, 2019). Results indicated that the number of studies related to diatom indices based on the ecoregion should increase to obtain clearer and more accurate results in Turkey.

As a result of the developed diatom indices, it is understood that pollution-sensitive and pollution tolerant species show the distribution in the different ecosystems having any of the five ecological conditions (high, good, moderate, poor, and bad). Incremental impacts of the stressors on biota lead to a decrease in the abundance of pollution-sensitive species but support pollution tolerant taxa. Considering that, these interactions are used to develop new diatom indices or improve them based on numerical data. Explaining multistressor interactions is not easy because it requires the use of multivariate statistical analyses. Therefore, multivariable statistical approaches are used to not only explain the relationship between diatom assemblages and environmental stressors but also evaluate the water quality of the ecosystem (Hering et al., 2010; Çelekli and Lekesiz, 2020; Freitas et al., 2020).

3.6. Trophic index Turkey

Even though bioassessment studies based on diatom metrics have been rapidly increased in Europe and given great importance, bioassessment studies of water bodies are still inadequate in Turkey. Therefore, the bioassessment studies based on diatom indices have been applied to evaluate the water quality of rivers in Turkey

Table 2. Developed diatom indices from different regions.

Index name	Abbr.	Origin	sp/genus	Reference
Percent community similarity of diatoms	PSc	USA	-	(Whittaker and Fairbanks, 1958)
Saprobic index	SI	Germany	-	(Zelinka and Marvan, 1961)
Water quality index	WQI	USA	-	(Horton, 1965)
Simple autecological index	SAI	USA	-	(Lowe, 1974)
Pollution tolerance index	PTI		-	(Lange-Bertalot, 1979)
Descy's pollution index	DES	France/Belgium	-	(Descy, 1979)
Diatom community index	DCI	Japan	117	(Sumita and Watanabe, 1983)
Acidity index	ACID	Sweden	-	(Henrikson and Medin, 1986)
Sládeček's index	SLA	Czechia	323	(Sládeček, 1986)
Leclercq and Maquet's index	LMI	France	210	(Leclercq and Maquet, 1987)
Diatom assemblages index for organic pollution	DAIpo	Japan	452	(Watanabe et al., 1986)
Generic diatom index	GDI	France	44 genera	(Rumeau and Coste, 1988)
European index	CEC	France	208	(Descy and Coste, 1991)
Pollution sensitivity index	PSI	France	4000	(Cemagref, 1982)
Steinberg and Schiefele's index	SHE	Germany	386	(Steinberg and Schiefele, 1988)
% of pollution tolerant taxa	%PT	Germany	-	(Schiefele and Kohmann, 1993)
Van Dam index	VDI	Netherlands	948	(Van Dam et al., 1994)
Trophic diatom index	TDI	England, Wales	177	(Kelly et al., 2008)
Biological diatom index	BDI	France	209	(Lenoir and Coste, 1996)
Artois-picardie diatom index	APDI	France	503	(Prygiel et al., 1996)
Eutrophication/pollution Index	EPI-D	Italy	222	(Dell'Uomo, 2004)
Percent aberrant diatoms	PAD	USA	-	(McFarland et al., 1997)
Rott trophic index	TI	Austria	650	(Rott et al., 1999)
Coring index	GM Seen	Germany	-	(Coring et al., 1999)
Behrendt and Opitz index	GM B&O	Germany	-	(Behrendt and Opitz, 1996)
Generic index	GI	Taiwan	161	(Wu, 1999)
Periphyton index of biotic integrity	PIBI	USA	38 genera	(Hill et al., 2000)
Pampean diatom index	PDI	Argentina	210	(Gómez and Licursi, 2001)
Swiss diatom index	DI-CH	Switzerland	708	(Buwal, 2002)
Generic diatom metric	IDG	France	11645	(Lecointe et al., 2003)
Diatom model affinity	DMA	USA	134	(Passy and Bode, 2004)
Swiss diatom index	SDI	Swiss	188	(Hurlimann and Niederhauser, 2006)
Diatom species index Australian Rivers	DSIAR	Australia	501	(Chessman et al., 2007)
Trophic diatom index for lakes	TDIL	Hungary	127	(Stenger-Kovács et al., 2007)
Ecological distance index	EDI	France	50	(Tison et al., 2008)
Eastern Canadian diatom index	IDEC	Canada	498	(Grenier et al., 2010)
Diatom multimetric index	MDIAT	Spain	18	(Delgado et al., 2010)
South African diatom index	SADI	South Africa	-	(Harding and Taylor, 2011)
Duero diatom index	DDI	Spain	137	(Álvarez-Blanco et al., 2013)
Trophic water quality index	TWQI	Brazil	70	(Lobo et al., 2015)
Richmond River diatom index	RRDI	Australia	142	(Oeding and Taffs, 2017)
Trophic index of Turkey	TIT	Turkey	219	(Çelekli et al., 2019b)
French Guiana diatomic index	FGDI	France	400	(Carayon et al., 2020)
Diatom ecological quality index	DEQI	Mexico	162	(Salinas-Camarillo et al., 2020)

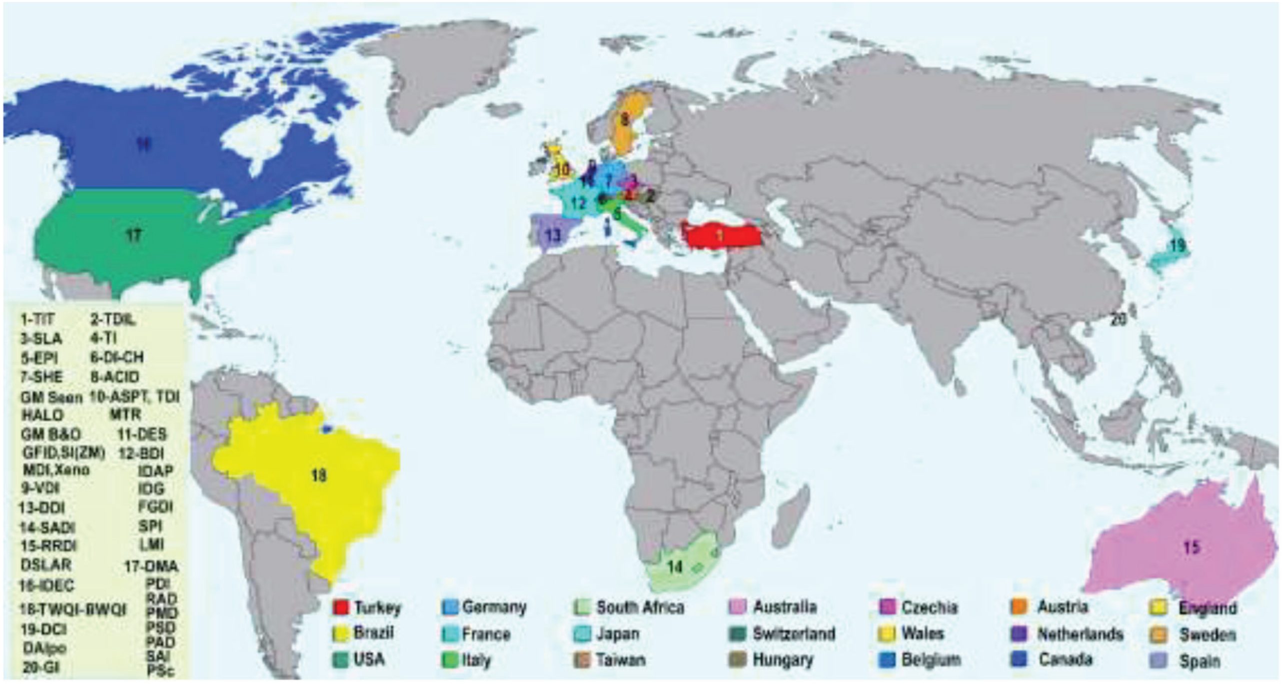

Figure 4. Diatom-based index developer countries. Full names of abbreviated diatom indices are given in Table 2.

but only in a few regions. A recently developed diatom index called TIT (trophic index Turkey) (Çelekli et al., 2019b) was the first diatom index in Turkey. The suitability of this index has been tested in different geographical and ecological regions of Turkey; for example, the freshwaters of the Western Anatolian basin (Toudjani et al., 2017), the North Aegean basin (Çelekli et al., 2018), Aras river basin (Çelekli et al., 2019a), the West Mediterranean basin (Çelekli and Lekesiz, 2020), and the southeast of Anatolia (Çelekli and Kapı, 2019; Çelekli and Arslanargun, 2019; Çelekli and Bilgi, 2019). Results indicated that TIT is an accurate diatom index when compared with the other diatom indices used to assess the ecological status of lotic ecosystems because of ecoregional environmental factors on the trophic weight of diatom taxa in Turkey (Lobo et al., 2004; Çelekli and Kapı, 2019). Turkey is one of the countries applying the WFD directive in the context of the EU integration process. Meeting the increasing demand for freshwater in Turkey will also be one of the major encountered problems in the future. Concerning that, studies dealing with the bioassessment of lotic ecosystems in Turkey have increased with a few important projects supported by the T.R. Ministry of Agriculture and Forestry, General Directorate of Water Management.

3.7. BCG-biological condition gradient

Aquatic ecosystems have different environmental factors in nature, and each ecosystem is affected by its unique stressors (Baert et al., 2016; Charles et al., 2021). Considering these reasons, it is difficult to evaluate each ecosystem with the same standard method.

These different reactions among biological quality components make bioassessment of the aquatic ecosystem difficult. Deciding how to set criteria for classifying water bodies in a biologically meaningful way is difficult (Milošević et al., 2020). The biological condition gradient (BCG) is an approach that will help to eliminate these challenges. The BCG is a comprehensive, descriptive, scientific, and ecosystem-based framework that describes a gradient in resource conditions including biological, physical, and chemical variables to standardize biological assessments of freshwater streams (Hausmann et al., 2016; Charles et al., 2019; Ruaro et al., 2020). The BCG was developed in the United States to standardize bioassessment in freshwater bodies with CWA's objectives. The BCG, a scientific characterization of the biological response to increasing effects of stressors, is an ecosystem-based framework that independently evaluates chemical, physical, and biological conditions (Davies and Jackson, 2006).

Ecological features reflecting the degree to which a system is moving away from its natural structure are expressed in the concept of biological status (Davies and Jackson, 2006; Hausmann et al., 2016; Charles et al., 2019; Ruaro et al., 2020). Six levels in the BCG have been briefly defined as level 1-natural or very little affected condition, level 2-minimal changes in biotic structure and ecosystem function, level 3-minimal changes in ecosystem function and significant changes in biotic community structure, level 4-moderate changes in biotic structure, minimal changes in ecosystem function, level 5-major changes in biotic community structure, moderate changes in ecosystem function, and level 6-biotic community dramatically changed, a great loss of ecosystem (Davies and Jackson, 2006; Hausmann et al., 2016).

As in the different ecoregions, the response of diatom species to environmental conditions is different at different pollution levels. Results of diatom indices will be more reliable in the bioassessment of the ecological status of water resources when the evaluation is supported by the results of BCG. Such studies will also allow the revision of developed diatom indices that will give more accurate results. However, the absence of diatom index studies using the BCG approach in the literature indicates that there is a gap in this regard. A significant effort has been put forward to evaluate water quality by using a combination of biological and chemical assessments. Results of BCG studies are used to evaluate environmental conditions of ecosystems (Hausman et al., 2016). The diatom indices-BCG evaluation is a useful approach to ensure reliable interpretation of water quality. Accordingly, the increase in such studies will give us the answers to the following questions: (i) Does an index developed according to the ecoregion give the same result in every level of BCG? (ii) Can a different index be used for each level of BCG? (iii) If a different index is used for each level of BCG, does it matter to develop the index with the ecoregion approach? and (iv) Will the indices to be used for each level of BCG give the similar results in different geographical regions? When we get the answers to these questions, a common method can be developed for monitoring studies using diatom indices all over the world, and this will allow us to interpret the results more accurately even in different regions. But for this, the number of diatom index studies supported by the BCG approach should be increased.

Consequently, independent, and different approaches have been developed to assess biological conditions and encourage new methods to interpret the conditions of aquatic ecosystems (USEPA, 2016). Therefore, the occurrence of a standard approach for assessing biological conditions will enable a common method and data exchange for scientists and many countries (Davies and Jackson, 2006). Considering the aforementioned information, BCG may be shown as an important standard method along with diatom indices to (i) biologically assess water quality, (ii) provide easier monitoring of high-quality water, and (iii) control the amount of water degradation (USEPA, 2016).

4. Conclusion

Aquatic ecosystems around the world are adversely affected by anthropogenic activities. Thereby, bioassessment of surface waters is becoming more important to accurately estimate the deterioration of ecosystems and to accomplish environmental sustainability according to the application of the WFD, which is critical and necessary for a holistic approach. This review is to emphasize the importance and advantages of using diatom metrics in the bioassessment of surface waters especially in running waters and the importance of indices developed based on ecoregions. Because using foreign diatom indices can lead to erroneous interpretation of water quality due to the ecoregional factors on the trophic weight of diatom taxa. Hence, many diatom indices are developed in different ecoregions of the world, but their numbers are not enough. Nowadays, the bioassessment of the ecological condition of inland water bodies based on diatom metrics is adopted worldwide. It is understood from the literature that diatoms are robust and reliable bioindicators for bioassessment studies, especially in running waters. Also, bioassessment should be supported by hydromorphological and physicochemical evaluations. In all, the present review suggests that biological, hydromorphological and physicochemical assessments should be supported by the biological condition gradient to accurately determine the water quality of surface waters.

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