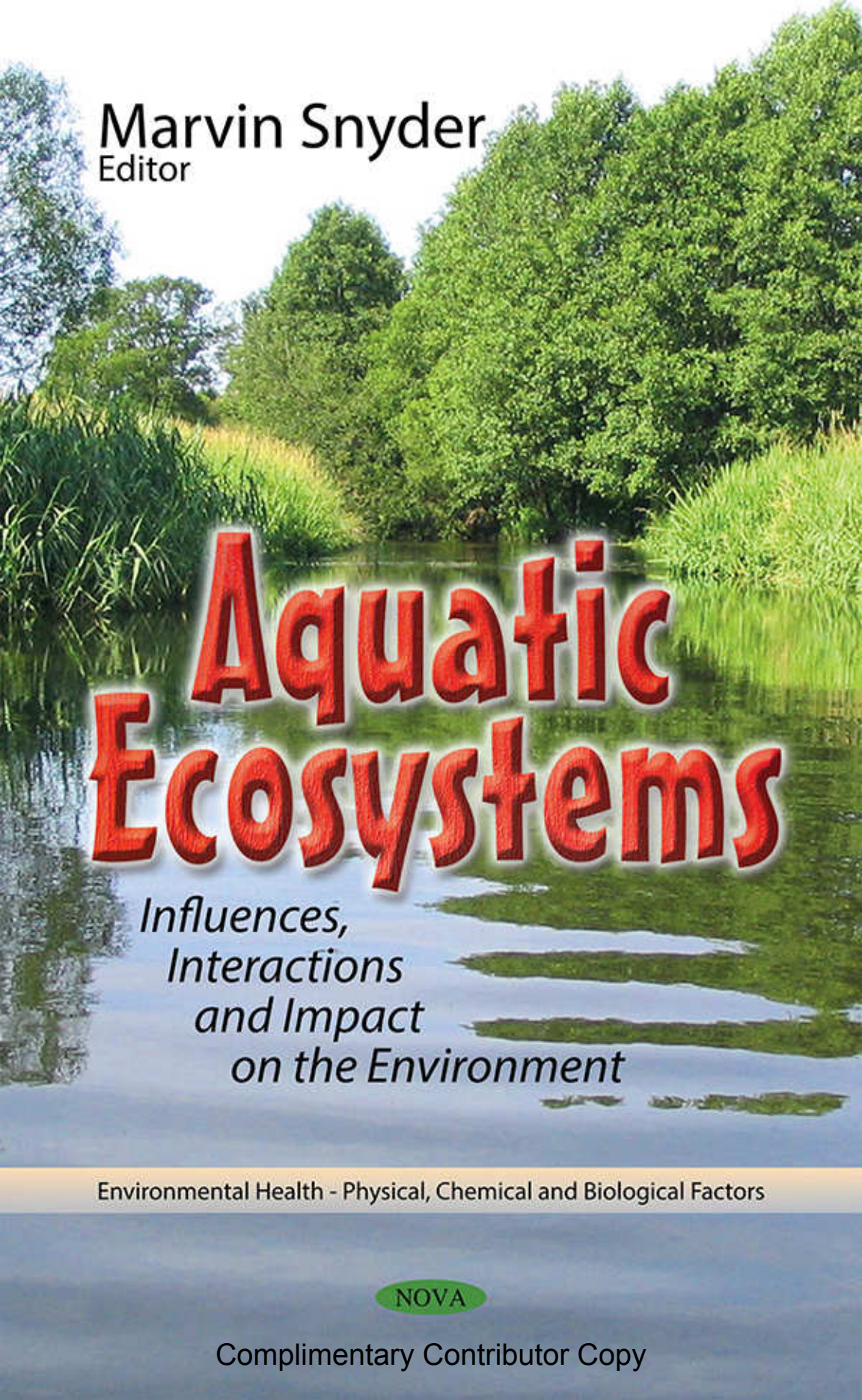




Marvin Snyder
Editor

Aquatic Ecosystems

*Influences,
Interactions
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on the Environment*

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MARVIN SNYDER
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Chapter 2

MICROALGAE AS WATER QUALITY INDICATORS: AN OVERVIEW

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ABSTRACT

Traditionally, the use of ecological indices has been supported by measurements and determinations of physical and chemical characteristics of the water when scientists were unable to detect damage in biological communities. In this context, a brief historical overview of biological indicators, considering the assessment of water quality through microalgae from an environmental perspective and their role as natural bioindicators of the effects of eutrophication, is presented. Indicators, indices and models have been developed for the study of water quality in freshwater, coastal brackish-water and marine environments. The indicators commonly used are concentrations of nutrients, dissolved oxygen, the elemental ratio of nutrients, and chlorophyll-*a*. Various indices composed of more than one variable have been developed to assess the trophic level of the study sites, and complex models integrate information from different compartments of the ecosystems based on

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physical and chemical variables, submerged aquatic vegetation, sediments and other factors. However, very few studies have considered phytoplankton associations as functional groups or as assemblages with different life strategies. This is one of the methodological lines to follow for the characterization of different types of water bodies.

Keywords: bioindicators, biological indices, ecological indices, eutrophication, life strategies, microalgae, phytoplankton, water quality

INTRODUCTION

The term “water quality” is not an absolute characteristic but rather an attribute socially defined in terms of the use of water resources (Revengea et al., 2000); every mode of water use requires a determined standard of quality. For this reason, to evaluate water quality, uses of the water should be considered (INE-SEMARNAT, 2005).

On the basis of the ecological criteria applied in Mexico to water quality, the latter is defined according to the parameters that constitute the minimum quality required for potable water, recreational activities, irrigation, livestock, aquaculture and aquatic wildlife conservation (SEDUE, 1989). However, ecological criteria of water quality imply that coastal waters of the whole country (Pacific, Gulf of Mexico and Caribbean coasts) are similar and do not consider advantages of the ecosystem health indicators used beyond Mexico, such as chlorophyll-*a*, harmful algal blooms and submerged aquatic vegetation (CNA, 1989; Herrera-Silveira, 2006).

Various indicators, indices and models have been developed to study eutrophication and water quality in freshwater, coastal (brackish-water) and marine environments (Escobedo-Urías, 2010). Nutrient and dissolved oxygen concentrations, elemental proportion in nutrients (N:P, Si:N, Si:P), chlorophyll-*a* and a diatom/flagellate ratio have been used more frequently (OECD, 1982; Vollenweider and Kerekes, 1982; EEA, 1999; Smayda, 2005; Escobedo-Urías, 2010), whereas diverse compound indices (with more than one variable) have been suggested for evaluation of trophic levels (Karydis et al., 1983; Ignatiades et al., 1992; Justic et al., 1995; Stefanou et al., 2000; Wasmund et al., 2002; Escobedo-Urías, 2010). Additionally, complex models that integrate information in different compartments of ecosystems have been proposed; these include physical and chemical variables, phytoplankton, zooplankton, benthos, submerged aquatic vegetation (macroalgae and

angiosperms) and sediments (NRC, 2000; Ferreira et al., 2009; Escobedo-Urías, 2010).

Traditionally, to determine water quality, physical-chemical parameters have been used, and the results obtained refer to the conditions in the short term (INE-SEMARNAT, 2000; De la Lanza-Espino and Gómez-Rojas, 2005), when discrepancies in the nitrogen and phosphorus concentration ranges are observed, and therefore, in trophic level (Aranda-Cirerol, 2004). The attributes of each ecosystem vary with the locality (Cloern, 2001; Aranda-Cirerol, 2004), which requires them to be related to the organisms due to their exposure to ambient conditions (De la Lanza-Espino and Gómez-Rojas, 2005).

However, this practice has resulted in serious disagreements between various investigation teams (Justic, 1991), because from the beginning the models developed for continental freshwater bodies were also applied to coastal waters without detailed evaluation due to discontinuity in the investigation (Cloern, 1999; Vidal et al., 1999; Escobedo-Urías, 2010). Some authors proposed indices and methodologies applicable to marine environments, based principally on nutrient concentrations, chlorophyll-*a*, transparency and/or dissolved oxygen (Karydis et al., 1983; Justic, 1991; Ignatiades et al., 1992; Vollenweider et al., 1998; Stefanou et al., 2000; Wasmund et al., 2002; Escobedo-Urías, 2010). This approach has been followed in North America (Plafkin et al., 1989; Resh et al., 1995; Barbour, 1997), Australia (Chessman, 1995, 2003; Schofield and Davies, 1996; Metzeling et al., 2002; Tiller and Metzeling, 2002), the United Kingdom (Wright et al., 1984, 2000), Africa (Chuter, 1972) and, in general, in European countries where methodologies within the European Union Water Framework Directive (WFD) were being adjusted (Bonada, 2003). Phytoplankton parameters are considered critical for water quality assessment within the WFD (Domingues et al., 2008; Wolfram et al., 2009; Spartharis and Tsirtsis, 2010), although not sufficient (WFD, 2000; Kaiblinger et al., 2009). Efficient tools for diagnostics (Hermosilla-Gómez, 2009; Anderson et al., 2012) and identification of key parameters will allow us to evaluate human impacts or ecological conditions (Hermosilla-Gómez, 2009).

A SHORT HISTORY OF THE BIOLOGICAL INDICATORS

The first efforts to determine ecological damage caused by domestic and industrial waste affecting water currents were performed by Kolkwitz and Marson (1902, 1908, 1909), thus establishing the basis for the saprobic system

widely used afterwards in Germany and some other European countries (Roldan-Pérez, 1999; Badii-Zabeh et al., 2005). During the early-mid-1950s, various methodological approaches to assess the water quality using biological indicators were implemented (Roldan-Pérez, 1999; Badii-Zabeh et al., 2005). Patrick (1949, 1950) proposed biological methods to evaluate ecological conditions of water currents. In the late 1950s and the early 1960s, the concept of species diversity based on mathematical indices derived mainly from information theory became a theme under discussion (Shannon and Weaver, 1949; Simpson, 1949; Brillouin, 1951; Margalef, 1951, 1956, 1958; Beck, 1955; Sheldon, 1969; Wilhm, 1970).

The use of ecological indicators that describe richness and diversity of species as indicators of contamination was introduced by a number of studies performed in the 1970s (Bechtel and Copeland, 1971; Archibald, 1972; Cook, 1976; McKenzie et al., 1992; Mann and Plummer, 1999; Ostrom et al., 1999; Smeets and Weterings, 1999; Tilman, 1999; Jackson et al., 2000; Cohen, 2003). These indicators give information on the structure of the biological communities (Pielou, 1984; Ludwig and Reynolds, 1988; Magurran, 2004), and can be seen to be influenced by environmental factors (Legendre and Legendre, 1983; Washington, 1984). The diversity indices proposed by Mihnea (1985) suggested an increase of the diversity values with diminishing productivity. Washington (1984) revised the indices with a special emphasis on aquatic ecosystems, presenting 18 diversity indices, 19 biotic indices and five similarity indices, and analyzing their applicability to biological systems (Roldan-Pérez, 1999; Badii-Zabeh et al., 2005). In Spain, Prat et al., (1986) compared two indices of water quality, one using physical-chemical variables (ISQA - Simplified Water Quality Index) and another using biological parameters (a family of indices applied in the Besos and Llobregat river basins, Spain, called FBILL or BILL), and found a low correlation between them.

Barbour et al., (1999) presented a total of 63 types of indices/measurements for a rapid assessment of ecosystems (Figure 1). Of them: 1) eight indices are based on the species richness measurements; 2) 15 indices are based on percentage of various species in terms of their abundance; 3) 15 indices correspond to community diversity and similarity indices, of which the most well-known are those of Shannon and Weaver (1949), Simpson (1949) and Margalef (1951); 4) 12 indices are referred to as biotic indices, including the Biological Monitoring Working Party (BMWP) and the Saprobity Index; 5) 10 indices are known as functional measurements where the role of the organisms in the community is considered (e.g., collectors, filtrators,

predators); 6) another three measurements are considered as combined indices: the Invertebrate Community Index (OEPA, 1989), the Mean Biometric Score (Shackelford, 1988), and the Biological Condition Score (Plafkin et al., 1989).

Karr (1991) introduced the concept of the index of biological integrity (IBI), which is a multi-parametric tool based on the fish community, to assess water currents. Armitage and Petts (1992) examined the possibility of the application of biotic scores and predictions based on a computerized system known as the River Invertebrate Prediction and Classification System (Figure 1) to evaluate the benthic fauna losses (Wright, 1995). Resh et al., (1995) developed rapid methods of assessment of water using aquatic macroinvertebrates as bioindicators. Alba-Tercedor (1996) adapted the use of aquatic macroinvertebrates in the programs of water quality evaluation in Spain. Townsend et al., (1997) assessed perturbation in the water environment in relation to macroinvertebrates and their species richness.

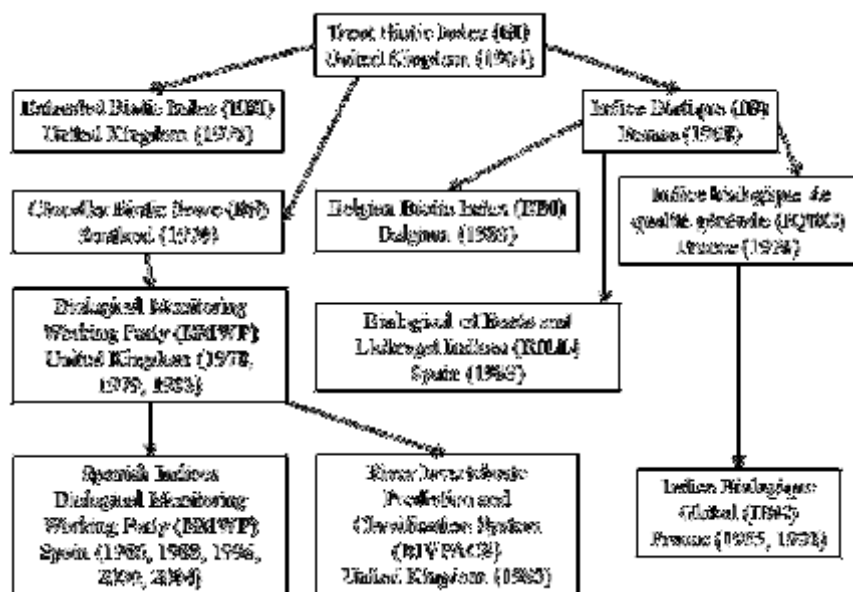


Figure 1. Biotic indices most common in Europe.

According to Cude (2001), the water quality indices are periodically reviewed, revealing new emphases and providing new tools for developing new indices (Dinius, 1987; Kung et al., 1992; Dojlido et al., 1994; Fernández-Parada et al., 2003). The first comparisons published elsewhere were those of

Landwehr and Deininger (1976) and Ott (1978). The latter reviewed the indices used in the USA; in addition, a detailed discussion about the use of the environmental indices in both theory and practice was given.

In Europe, van Helmond and Breukel (1996) showed that on a global scale at least 30 water quality indices were commonly used, with a number of variables ranging between 3 and 72, of which at least the following three were frequently used: molecular oxygen (O_2), biochemical oxygen demand (BOD) or chemical oxygen demand (COD), ammonium (NH_4-N), orthophosphates (PO_4-P), nitrates (NO_3-N), hydrogen potential (pH), and total solids (TS) (Fernández-Parada et al., 2003). Similarly, in Croatia, Stambuk-Giljanovic (1999) observed that through the years, various water quality indices were formulated with their own objectives. In South Africa and Australia, Cooper et al., (1994) and Richardson (1997) reviewed the indices to elaborate their own indices for estuaries.

To classify coastal waters into oligotrophic, mesotrophic and eutrophic, many indices were proposed, having in common a capacity to sum up the data obtained in the field (Karydis, 2009). Abiotic indices based on nutrient values are widely used; the N:P ratio in a given water body was applied to define eutrophic conditions (Ryther and Dunstan, 1971). To define a trophic state based on nutrient content, certain algorithms were designed (Karydis et al., 1983), and nutrient concentration variations were suggested as indicators of trophic conditions (Giovannardi and Tromellini, 1992; Ignatiades et al., 1992; Stefanou et al., 2000).

MICROALGAE AS INDICATORS

Microalgae are natural bioindicators of eutrophication due to their rapid response to fluctuations of environmental conditions (Livingston, 2001) resulting from the intensity and the type of human activities (Harding, 1994; Marcela, 2003; Kelly-Gerreyn et al., 2004; Álvarez-Góngora and Herrera-Silveira, 2006).

On a global scale, most of the studies based on phytoplankton communities for environmental assessment were performed in lakes (rarely involving phytobenthos), due to the ease of sampling microalgal biomass and because of the defined ecological preferences of individual planktonic species. Phytoplankton provides information on environmental conditions in general (Table 1), and it serves as a base to extend the classification of lakes in relation to water quality and, in particular, their trophic state (Salmaso et al., 2006;

Bellinger and Sigeo, 2010). Rivers and flowing watercourses in general have also been objects for monitoring water quality (Kelly and Whitton, 1995; Whitton and Rott, 1996; Semenchenko, 2004). Microalgae are common dwellers of aquatic environments: freshwater, brackish-water and marine, also inhabiting rocks, macroalgae or submerged surfaces (Darley, 1987), where both planktonic and microphytobenthic assemblages are used for characterization of aquatic habitats with the use of biological, physical-chemical or hydromorphological indicators (Hermosilla-Gómez, 2009).

Table 1. Attribute indicators of phytoplankton and phytobenthos in water bodies (Confederación Hidrográfica del Ebro, 2001, 2005)

Phytoplankton	Phytobenthos
Suitable for detection and monitoring of the physical-chemical pressure related to: • Thermal pollution. • Changes in water mineralization (and composition of dissolved major ions). • Eutrophication (nitrogen and phosphorus concentrations and sometimes silica and other cations such as iron). • Organic pollution (soluble and particulate).	Respond to increased nutrient (mainly P and N) concentrations in water by changing their composition; in some cases, involves a decrease in biodiversity and increase in biomass, so that when the water becomes eutrophic, substrates are coated with films of green or brown algae.
Phytoplankton is indicative of the hydromorphological pressure (alterations of the hydrological regime and morphological conditions of the seabed) for determining changes in the rate of renewal of lakes and reservoirs.	They are insensitive to the hydromorphological pressure (alterations of the hydrological regime and morphological conditions of the seabed), so their use for the detection of these pressures is not recommended.

Kane (2004) reviewed the main ecological indices, particularly those applied to lakes. He referred to Lindeman (1941), who designed the framework for trophic dynamic studies; to Shannon and Weaver (1949) and Wiener (1948) who developed indices for species diversity measurements and measurements of the potential number of interactions between taxa; to limnologists who developed indices for the use of individual parameters, such

as oxygen loss (Hutchinson, 1957), primary production (Rodhe, 1958), total phosphorus (Chapra and Robertson, 1977), total nitrogen (Vollenweider, 1968), chlorophyll-*a* (Dobson et al., 1974), and turbidity based on Secchi disc measurements (Vallentyne et al., 1968).

Determination of chlorophyll-*a* in water, which is closely related to the process of primary production, is one of the most common methods to quantify indirectly phytoplankton biomass (Contreras-Espinosa et al., 1994). This method has been used since the 1960s and has contributed valuable information on the presence and abundance of phytoplankton species (Magaña-Álvarez, 2004).

Later it was found that chlorophyll-*a* concentration was not as closely and directly related to primary production as had been previously suggested (Nixon, 1988). The following possible reasons were proposed for that: grade of maturity of phytoplankton community, physiological stage, amount of phaeopigments and accessory pigments (Margalef, 1974), and abundance of nanoplankton (<20 μm), which plays an important role earlier neglected (Magaña-Álvarez, 2004). Various studies have proven a significant contribution (up to 90%) of nanophytoplankton to the total phytoplankton biomass and primary production (Malone, 1971; McCarthy et al., 1974; Gilmartin and Revelante, 1978; Platt et al., 1983; Glover et al., 1985; Hopcroft and Roff, 1990; Lara-Lara and Bazán-Guzmán, 2005). Also, it has been shown that, in general, the nanophytoplankton fraction dominates in oceanic waters and microphytoplankton (20 to 200 μm) in neritic waters (Malone, 1980; Cole et al., 1986; Harris et al., 1991; Lara-Lara and Bazán-Guzmán, 2005).

Several microalgal cells or major taxa also serve as indicators of trophic state in coastal waters (Kitsiou and Karydis, 1998), where their use for assessment of water quality has a long history (Padisák et al., 2006) of applying species indicators of saprobic conditions (Thunmark, 1945; Nygaard, 1949; Järnefelt, 1952; Teiling, 1955; Heinonen, 1980; Rosén, 1981; Kümmerlin, 1990; Gulyás, 1998; Lepistö and Rosenström, 1998; Lepistö, 1999). Cell abundance is influenced by intrinsic factors (biological factors, plankton sedimentation rate, turbidity/stability of water, residence time of water, nutrients, organic matter, mineralization, pH and oligoelements (Valiela, 1984; Confederación Hidrográfica del Ebro, 2001).

Numerous ecological indices were developed as described by Thunmark (1945), Nygaard (1949), Hörnström (1981), Brettum (1989), Tremel (1996) and Schönfelder (1997). However, none of them has been widely applied due to the characteristics of the dynamics of phytoplankton communities, diversity of habitats in freshwater ecosystems and geographic differences (Padisák et

al., 2006). Domingues et al., (2008) analyzed constraints on the use of phytoplankton in Portuguese waters within the WFD, mentioning the absence of historical data for some water bodies, the unsatisfactory sampling frequency, the overlooking of blooms of pico- and nanoplankton and overestimation of large microphytoplankton, the impossibility (in many cases) of discrimination between autotrophic and heterotrophic cells, and the necessity of implementation of remote sensing and chemo-taxonomical analyses in the phytoplankton monitoring programs. In Greece, Spartharis and Tsirtsis (2010) analyzed the efficiency of 22 ecological indices, selected the most efficient of them (Menhinick, 1964) diversity index and three indices of evenness), and proposed an Integrated Phytoplankton Index (IPI) based on chlorophyll-*a*, cell abundance and diversity, which seems to be advantageous as one of the synthetic indices.

LIFE STRATEGIES AND FUNCTIONAL GROUPS OF PHYTOPLANKTON

Reynolds et al., (2002) published an extensive description of phytoplankton associations as functional groups, with the groups of species related to different combinations of physical, chemical and biological properties of lake environments (depth of the mixing layer, light, temperature, phosphorus, nitrogen, silicon, carbon dioxide and grazing pressure). This concept is closely related to the concept of primary life strategies developed on the basis of terrestrial plant communities, of which two principal schemes can be distinguished: those with two strategies (MacArthur and Wilson, 1967; Pianka, 1970) and those with three (Ramensky, 1938; Grime, 1979; Rabotnov, 1978). Previously, a life strategy was understood as an assemblage of adaptations that provides the possibility for a species to co-exist with other organisms and to occupy a certain status in a biocoenosis (Rabotnov, 1975). Among the first category, the concept of *r*- and *K*-selection based on a logistic growth model (MacArthur and Wilson, 1967) is the most popular. Pianka (1970) considers them two extremes and visualizes it as an *r/K*-continuum. Other authors also followed the paradigm of a continuum for three- and five-type models (Mirkin, 1983; Romanovsky, 1989a, b).

The CSR model (C - competitors, S - stress-tolerators, and R - ruderals), based on stress and disturbance (biomass destruction) as determining factors, is an example of the second category (Grime, 1979). Ramensky's (1938) three

types, or plant coenotypes, or phytocoenotypes (V - violent, or “lions,” P - patient, or “camels,” and E - explerent, or “jackals”), in many respects correspond to those interpreted as dominant, enduring and explorative, respectively, although they are not identical to Grime’s types (Onipchenko et al., 1998), and the coenotic significance of each species may change from one coenosis to another. Patients correspond to *K*-species and violent and explerents to *r*-species (Romanovsky, 1989a). Romanovsky (1989a, b, 1998), based principally on the studies of freshwater cladocerans, presented arguments in favor of a “triangular” shape of life strategies, emphasizing the differences in competitiveness between *r*-species. Referring to a three-type model also as to Ramensky-Grime classification, almost “forgotten” in international literature, Romanovsky (1989b) stresses the uselessness of ascribing a certain type of life strategy to a given species, stating that it makes sense only when at least two closely related populations are compared. It was shown that the amplitude and frequency of fluctuations of limiting resource dynamics play the utmost importance in life-history evolution (Romanovsky, 1989b); according to this view, a life-history strategy can be defined as “a combination of absolute and relative costs of incoming energy in survival and reproduction processes of population or a part of it, resulting in the maximal effectiveness of consumption of limiting resource at a certain regime of fluctuation of its abundance” (Romanovsky, 1989b: 30).

Mirkin (1983), based on Grime-Ramensky classification, Rabotnov’s works and Whittaker (1975), distinguishes five primary life-strategy types: violent, ecological and coenotical patients, and true and false explerents; in addition, transitional (secondary) types are recognized.

In the beginning, only the *r/K*-classification was used in phytoplankton studies (Margalef, 1978). However, in the late 1990s the CSR model was also applied, in particular, to the marine Subarctic phytoplankton community of Kandalaksha Bay of the White Sea (Ilyash, 1998; Ilyash et al., 1999). For the first time, it was concluded that the CSR theory could be successfully used for organisms (microalgae) lacking a pronounced age structure, with a fluctuating regime of mineral resources being the determining factor; the adaptive mechanisms that determine the life-strategy type were revealed. The annual phytoplankton cycle in high latitudes and temperate waters was shown to be a succession of violent-patient assemblages against the background of constant vegetation of patient generalists and explerents. It should be underlined that at present, our knowledge of metabolic interactions between microalgal species and thus investigations on sensitivity to competition among taxa, are limited,

to a greater extent, to cultivated species and remain a challenge for future studies.

CONCLUSION

Recently, a great number of environmental agencies of the world have applied methods of assessment of water quality based on biological communities (Barbour et al., 1999; Ospina-Álvarez and Peña, 2004). However, only a few of them have considered phytoplankton associations as functional groups, which constitutes one of methodological lines to characterize different types of lakes, lagoons and reservoirs, and to determine the variables necessary to evaluate their ecological state (Confederación Hidrográfica del Ebro, 2001). The CSR theory has shown that microalgae with different primary strategies also differ in survival strategies in relation to toxicants (Ilyash, 1998), so its application might enrich the water quality studies.

To identify species associations requires scientist to obtain lists of microalgae inhabiting lakes, lagoons or reservoirs and analyze their patterns of variation in species composition using statistical techniques (e.g., principal component analysis or agglomerative clustering). In the next stage, community indices can be elaborated for each type of water body (Reynolds et al., 2002). Thus, the information obtained via ecological indicators can also be used to predict future changes in the environment to identify action plans for a remedy or to follow tendencies in the indicators themselves (Niemi and McDonald, 2004). Understanding the life strategy of a microalgal species, its population or its part determined principally by the amplitude and frequency of fluctuations of limiting resource dynamics, will help in facing the problems related to harmful algal blooms that constitute only a part of a more challenging task - to control and to manage water quality in marine, brackish and freshwater environments.

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