Algae as indicators of environmental change

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Abstract

Despite an increased awareness by governments and the general public of the need for protecting all types of aquatic habitats, human impacts continue to impair the services that these ecosystems provide. Increased monitoring activities that focus on all major biological compartments are needed to quantify the present condition of Earth's aquatic resources and to evaluate the effectiveness of regulations designed to rehabilitate damaged ecosystems. Algae are an ecologically important group in most aquatic ecosystems but are often ignored as indicators of aquatic ecosystem change. We attribute this situation both to an underappreciation of the utility of algal indicators among non- phycologists and to a lack of standardized methods for monitoring with algae.

Because of their nutritional needs and their position at the base of aquatic foodwebs, algal indicators provide relatively unique information concerning ecosystem condition compared with commonly used animal indicators. Algae respond rapidly and predictably to a wide range of pollutants and, thus, provide potentially useful early warning signals of deteriorating conditions and the possible causes. Algal assemblages provide one of the few benchmarks for establishing historical water quality conditions and for characterizing the minimally impacted biological condition of many disturbed ecosystems. Preliminary comparisons suggest that algal indicators are a cost-effective monitoring tool as well.

Based on available evidence from field studies, we recommend development of taxonomic indicators based on diatoms (Bacillariophyceae) as a standardized protocol for monitoring ecosystem change. Both population- and community-level indices have inherent strengths, and limitations and information from both levels of biological organization should be utilized in tandem. However, further information concerning species tolerances to a variety of anthropogenic stressors is needed if autecological indices are to be used routinely for monitoring purposes. While functional measures (e.g. productivity) may also prove useful as monitoring tools, further investigation is required to characterize the reliability of alternative methodologies and to assess the consistency of these indicators under varying field conditions.

Introduction

While increased regulatory efforts in the United States and elsewhere have yielded many improvements in environmental quality compared to only a few decades ago (e.g. Gameson & Wheeler, 1977; Smith *et al.*, 1987), environmental degradation persists on a glob-

al scale and broad environmental goals articulated by many governments in recent decades have yet to be achieved. Since the passage of the amended Federal Water Pollution Control Act (FWPCA) in 1972, for example, certain aspects of water quality in U.S. rivers have declined (e.g. increased nitrate and cadmium loading) while other conditions (e.g. lead con-

centrations) have improved (Smith *et al.*, 1987). In the largest North American watershed, the Mississippi Basin, inputs of toxins, nutrients, and sediment from nonpoint sources continue to exert substantial impact on fisheries and other ecological resources, and, consequently, many tributaries fail to support their designated uses (Holland-Bartels, 1992). On a global scale, losses of aquatic habitat and biodiversity mount in response to the cumulative impact of chemical pollution, as well as other stressors including physical modifications and exotic species (Wetzel, 1992; Allan & Flecker, 1993). Emerging issues, such as the effects of global climate change on aquatic communities and ecosystems (Carpenter *et al.*, 1992), complicate predictions of future trends for freshwater resources.

Failures of regulatory policy to achieve stated environmental goals are due in part to insufficiencies in the existing capability to predict ecosystem risk accurately and to implement corrective action when unacceptable damage occurs (Cairns & Niederlehner, 1992; Knopman & Smith, 1993). Traditional approaches to risk assessment, which rely inordinately on information from laboratory bioassays, suffer from a number of shortcomings, including the failure to validate laboratory results under field conditions such that errors in prediction and resulting regulatory decisions are not corrected. To rectify this situation, both government agencies and non-government groups are expanding surveillance and monitoring programs in order to evaluate resource conditions and trends directly.

In an effort to characterize more accurately the cumulative impact of human activities on ecosystems, environmental monitoring has shifted away from a sole reliance on chemical indicators and towards the increasing use of measures of ecological condition. Numerous criteria have been proposed for selecting biological indicators based on such disparate concerns as the scientific objectives of the monitoring program, the availability of suitable methods and trained personnel, and the amount of public concern about different taxa. Algae have historically been an important component of biological monitoring programs. However, many of the recent advances in methods and indices for monitoring have been with other groups such as fish and macroinvertebrates (e.g. Karr *et al.*, 1986; Plafkin *et al.*, 1989). A lack of equivalent advances (e.g., standardized protocols for sampling and interpretation) for algae has undoubtedly made this group less attractive for regulatory use. Here, we review the merits of algae as environmental indicators based on commonly accepted criteria for indicator selection and

compare the efficacy of different types of algal indicators for monitoring ecosystem change in order to further the development of standard protocols for this group.

Criteria for indicator selection

Although a compelling ecological argument can be made for including algal indicators in environmental monitoring and assessment programs, a number of other factors must be weighed when evaluating both the relative utility of algal indicators compared with other taxonomic groups and the relative merit of different types of algal indicators. Many useful algal indicators have been used to assess environmental status and trends and others show promise for future development (see Table 1). However, human and financial resource constraints dictate that only a few be measured in any single monitoring program. Numerous criteria have been proposed for selecting the most suitable indicators for different purposes. In a synthesis of earlier views on the subject, Cairns *et al.* (1993) proposed a list of generic attributes for biological indicators. Ideally, an indicator should be (from Cairns *et al.*, 1993):

1. *biologically relevant i.e.* easily related to the maintenance of ecological integrity (*sensu* Karr, 1993);
2. *socially relevant i.e.* of obvious value to those involved in the decisionmaking process, including the general public;
3. *broadly applicable* to many stressors and sites;
4. *sensitive* to stressors, preferably without an all or none response or excessive natural variability;
5. *measurable*, in that it can be operationally defined and quantified using an accepted procedure with known precision and accuracy;
6. *interpretable i.e.* capable of distinguishing acceptable from unacceptable conditions in a manner that is scientifically and legally defensible;
7. capable of *continuity* of measurement *through time and space*;
8. of an *appropriate spatial and temporal scale* for the assessment under study;
9. *not redundant* with other measures included in the monitoring program;
10. *integrative* by summarizing information from many other possible indicators that cannot be feasibly measured;
11. *anticipatory i.e.* capable of providing a signal of ecosystem deterioration before serious harm has occurred;

Table 1. Algal attributes and associated indicators commonly employed in biological surveys and monitoring programs. Additional indicators which show promise for future use and are currently under development by various investigators are also included. Citations listed are intended to provide methodological illustrations of indicator use and efficacy.

Attribute	Indicator/Method	Illustrative citations
COMMUNITY STRUCTURE		
Biomass	Ash-free-dry-mass	Biggs, 1989
	Cell biovolume	Stevenson & Lowe, 1986
		Williams, 1972
	Chlorophyll *a*	Amblard *et al.*, 1990
		Biggs, 1989
		Delong & Brusven, 1992
		Lalonde & Downing, 1991
	Spectral reflectance	Mittenzwey *et al.*, 1988
Size spectra	Cell size distribution	Biggs *et al.*, 1978
		Sprules & Munawar, 1986
Diversity	Species diversity	Amblard *et al.*, 1990
		Archibald, 1972
		Marcus, 1980
		Patrick *et al.*, 1954
		Raschke, 1993
		Stevenson, 1984
	Species evenness	Marcus, 1980
	Species richness	Amblard *et al.*, 1990
		Raschke, 1993
Composition	Multivariate analyses	Agbeti, 1992
		Dixit *et al.*, 1989
		Guzkowska & Gasse, 1990
		Sabater *et al.*, 1991
		Swift & Nicholas, 1987
	Similarity indices	Marshall & Mellinger, 1980
		Raschke, 1993
		Stevenson, 1984
COMMUNITY METABOLISM		
Net Production	Change in biomass	Blum, 1956
		Sládeček & Sládečková, 1964
	Relative specific growth rate	Biggs, 1990
Productivity	Oxygen evolution	Crossey & LaPoint, 1988
		Kosinski & Merkle, 1984
		Uehlinger & Brock, 1991
	Radioisotopic tracer (^{14}C)	Lewis, 1992
		Munawar *et al.*, 1986
Photosynthetic Capacity	In vivo fluorescence	Goutx *et al.*, 1987

Table 1 (continued.).

Attribute	Indicator/Method	Illustrative citations
Bioaccumulation	Nutrients	Grimshaw *et al.*, 1993
		Lyngby, 1990
	Metals	Foster, 1982a
		Ramelow *et al.*, 1992
		Whitton *et al.*, 1989
Metabolic State	Adenylate energy charge	Fitzwater *et al.*, 1983
		Hino, 1988
Biomolecules	Ribonucleic acid	de Madariaga & Joint, 1992
Enzyme activity	Alkaline phosphatase activity	St Amand *et al.*, 1989
		Vincent, 1981
POPULATION ANALYSES		
Indicator Species	pH index	Battarbee *et al.*, 1986
		Cumming *et al.*, 1992
		Whitmore, 1989
	Pollution tolerance index	Descy, 1979
		Lange-Bertalot, 1979
		Palmer, 1969
		Watanabe, 1962
	Saprobien index	Fjerdingstad, 1964
		Kolkwitz & Marsson, 1908
		Pantle & Buck, 1955
		Sládeček, 1973
	Trophic index	Agbeti & Dickman, 1989
		Whitmore, 1989
Growth	Algal growth potential	Dodds & Randel, 1992
		Walsh, 1988
	In situ 'cages'	Eide *et al.*, 1979
		Munawar & Munawar, 1987

12. *timely i.e.* capable of providing information rapidly enough that management actions can be implemented before unacceptable damage occurs;

13. *diagnostic* of the particular stressor causing the problem;

14. *cost-effective* by providing the maximum amount of information per unit effort;

15. one for which an historical database exists in order to define a normal operating range (*i.e.* a range of acceptable conditions) and detect trends in ecosystem condition; and

16. *nondestructive* to the ecosystem.

Clearly, no single measure will possess all these attributes, and, indeed, some of the attributes are explicitly incompatible (e.g. 3 and 13). Different types of information may be required depending on the pur-

pose and scope of a monitoring program, and no one set of attributes makes an indicator ideal for all purposes as discussed by Cairns *et al.* (1993). Here, we consider each criterion separately in order to evaluate the general utility of algae as environmental indicators rather than the use for different indicators for specific purposes.

Evaluating algal parameters as indicators of environmental change

Biological and socioeconomic relevance

Algae are the dominant primary producers in most freshwater and marine environments and, thus, pro-

vide the principal energy base for many aquatic food-webs. Even in ecosystems where allochthonous energy inputs are considerable (e.g. headwater streams), autochthonous algal production can still contribute significantly to ecosystem energetics (e.g. Mayer & Likens, 1987). Algae play important biogeochemical roles in terms of nutrient fixation and spiraling within the ecosystem. Macroalgae also provide both a breeding habitat and a refugium for various animals (Boaden et al., 1975; Power, 1990). Thus, just as the condition of woody plants and other conspicuous vegetation is a focal point of terrestrial monitoring programs (Riitters et al., 1990), so too should the assessment of the composition and productivity of the algal assemblage be an integral part of aquatic monitoring programs.

Evaluations of algal production often focus on estimates of quantity, such as primary productivity and standing crop, and ignore the strong influence that changes in the quality of algal production can have on food web interactions. Algal taxa vary greatly in their edibility, and shifts in species composition can affect feeding relationships, population growth, and guild structure at higher trophic levels in aquatic food webs (Porter, 1976; Browder, 1982; Lamberti & Moore, 1986; Gilbert, 1990; DeMott & Moxter, 1991). Consideration of shifts in the taxonomic composition as well as the productivity of the algal assemblage in response to anthropogenic disturbances is often required, therefore, in order to predict accurately the effects on other ecosystem compartments.

Changes in the quantity and quality of algal production in response to increased nutrient loading can severely damage commercial and recreational fisheries. The dramatic influence of excessive algal production on ecosystem conditions (e.g. anoxia, fish kills) resulting from the increased loading of limiting nutrients such as phosphorus has been well documented (Edmondson, 1961; Hynes, 1969; Leach et al., 1977; Saether, 1980). Water quality managers are frequently concerned with the effect that blooms of nuisance algae have on the taste and odor of water in municipal water supplies. Excessive growth of algae can reduce the aesthetic appearance of lakes and swimming beaches. In the Laurentian Great Lakes, for example, Cladophora blooms significantly impair the beneficial uses of the shoreline (e.g. public beaches) (Auer et al., 1982). Conditions that favor the growth of toxin-producing algae can severely impact not only aquatic species, such as fish, but livestock and other animals as well (Gorham, 1964).

Range of applicability

The algal assemblage is a conspicuous component of virtually all aquatic habitats and many taxa are globally distributed. Although the extent to which algae and other microbes exhibit a cosmopolitan distribution is unclear (Cairns, 1991), algal species typically exhibit wider distributions among ecosystems and geographical regions than most higher organisms. Individual species exhibit little variation in autecology within their distributional range such that similar species assemblages occur predictably at locations having comparable physical and chemical conditions (Chessman, 1986; Sabater & Roca, 1990; Vyverman, 1992; Blinn, 1993). These attributes reduce problems of standardizing indices across watersheds or other geographical units, such as those encountered for fish communities (Karr et al., 1986; Leonard & Orth, 1986; Oberdorff & Hughes, 1992).

Use of large organisms such as fish to detect environmental change can be problematic in low-order streams where reference communities are represented by only a few species. Algal assemblages are typically species rich even in headwater streams and, in the absence of anthropogenic disturbance, do not exhibit the marked longitudinal fluctuations in community structure that have been documented for other aquatic groups. For example, Molloy (1992) found few significant changes in diatom species composition on standardized artificial substrata among first- to fifth-order reaches of streams in Kentucky, USA. Therefore, it should be possible to establish a reference range for some algal indicators that is constant, or at least reasonably predictable, with changes in stream size. Such consistency across longitudinal gradients is attractive as it is often difficult to find larger streams that can serve as reference sites for characterizing the minimally-impacted ecological state.

The use of algal indicators of ecosystem condition has historically focused on effects associated with organic enrichment and other forms of cultural eutrophication (Patrick, 1977). Early efforts at developing algal indicators focused on refining and expanding the saprobien system introduced by Kolkwitz & Marsson (1908). Saprobien indices have limited applicability to contemporary pollution problems such as increased loadings of toxic chemicals. However, laboratory and field evidence indicates that algae exhibit predictable responses to a variety of environmental stressors, although the nature of this response may vary depending on the type of impact. Both commu-

nity structure and function are affected by a variety of chemicals at environmentally realistic concentrations, including pesticides (Walsh, 1972; Hawxby *et al.*, 1977; Kosinski, 1984; Krieger *et al.*, 1988), heavy metals (Sigmon *et al.*, 1977, Thomas & Seibert, 1977; Pratt *et al.* 1987; Pratt & Bowers, 1990; Scanferlato & Cairns, 1990), PCB's (Harding, 1976), and complex effluents (Amblard *et al.*, 1990). Algal surveys have been useful in understanding the effects of multiple environmental changes associated with acidification in lakes in the northeastern United States (e.g., Stevenson *et al.*, 1985, Dixit *et al.*, 1989; Cumming *et al.*, 1992). While considerable work is still required to develop reliable algal indices of different forms of chemical stress, these previous studies indicate that such efforts are warranted.

Most surface waters are subject to simultaneous impact from multiple environmental stressors, usually including some form of nutrient enrichment. In order to integrate the effects of multiple stressors into an unbiased expression of cumulative impact, biological indicators should not be unduly sensitive to one type of stressor. Algal production in aquatic ecosystems is often constrained by one or a few macronutrients (Hansson, 1992) and increased nutrient loading may substantially alter algal responses to other inputs. Basin-wide changes in the algal flora of the Laurentian Great Lakes, which have been subjected to numerous biological, chemical, and physical insults over the past century, have been linked primarily to increased nutrient loading (Stoermer, 1978; Wolin *et al.*, 1988). Increased availability of limiting nutrients can significantly reduce algal community sensitivity to certain chemical stressors (e.g. Chen, 1989; Wängberg & Blanck, 1990). Given the close correspondence between algal condition and trophic status, further consideration of the interactive effects of nutrient enrichment and other stressors on algal indicators is certainly needed.

Sensitivity

There is no compelling evidence that algae are consistently more sensitive to toxic chemicals than other aquatic organisms; rather, available evidence indicates that tolerances among major phyla and kingdoms are quite similar. For example, the sensitivity of a diatom species to a variety of common chemicals in industrial effluents was not consistently greater or less than that of snail and fish species (Patrick *et al.*, 1968). However, controlled laboratory and field studies have shown that common toxic pollutants elicit responses from algae and other protists at concentrations at or below those deemed 'safe' for aquatic life on the basis of standard invertebrate and vertebrate bioassays (Genter *et al.*, 1987; Pratt *et al.*, 1987; Niederlehner & Cairns, 1990). Morgan (1972) compared the toxicity of a PCB and the pesticide DDT to freshwater organisms representing several trophic levels and found that, while algal species were most sensitive to short-term exposure to these compounds, invertebrates were more sensitive to chronic exposures. The alga *Chlorella* was found to be considerably less sensitive to most toxicants compared with standard test animals such as *Daphnia* (Kenaga & Moolenaar, 1979). This last assessment must be interpreted cautiously as coccoid green algae tend to be extremely pollution tolerant (e.g. Genter *et al.*, 1987), whereas *Daphnia* is routinely used in chemical testing precisely because it tends to be extremely sensitive to many chemicals.

As with other groups of organisms, the sensitivity of algae to a given stressor may vary markedly in relation to other environmental factors. The response of algal growth and metabolism to heavy metals varied considerably depending on the presence of natural chelating agents (e.g., humic acids), nutrient concentration, as well as other abiotic and biotic parameters (e.g., Bentley-Mowat & Reid, 1977). Prior exposure to environmental contaminants can alter the sensitivity of protist assemblages to a novel stressor (Niederlehner & Cairns, 1993). Populations may exhibit decreased sensitivity or adaptation to a stressor as a result of chronic exposure (Foster, 1982a, b; Blanck & Wängberg, 1988). Sensitivity is thus determined by several environmental factors including the nature and duration of exposure.

Substantial interspecific variation in sensitivity to chemical stressors exists within major animal groups routinely used for testing and monitoring (e.g., Mayer & Ellersieck, 1986). Algal taxa have also been found to vary considerably in their sensitivity to common pollutants (Hollister & Walsh, 1973; Blanck *et al.*, 1984; Genter *et al.*, 1987). Although there is evidence that some species of diatoms may be generally 'resistant' to pollution whereas others may be generally 'sensitive' (e.g., Lange-Bertalot, 1979), critical tests of this hypothesis for different types of stressors have yet to be published in the literature. Furthermore, individual populations may fluctuate dramatically in response to changes in environmental conditions unrelated to anthropogenic impacts. Given the problems associated with consistently predicting the sensitivity and

response of individual species to chemical stress (see Cairns & Niederlehner, 1987), the use of community-wide assessments, which are relatively insensitive to the erratic behavior of a few populations, should provide the most reliable indications of ecosystem condition.

Measurability

The numerous algal parameters proposed for assessing environmental quality vary greatly in their ability to be quantified using a generally accepted protocol with minimal error. Although a conceptual debate on the relative merits of structural and functional measures for evaluating ecosystem stress persists (e.g. Cairns & Pratt, 1986; Crossey & LaPoint, 1988), structural indicators are generally easier to standardize and measure under field conditions. The measurement of functional properties often suffers from a lack of generally accepted or standardized procedures and many protocols require excessive field incubation time (e.g., hours), which can severely limit the number of stations sampled in a single work day. This problem has been minimized by the use of shipboard or laboratory incubations to estimate production capacity, although simulating natural light conditions in the laboratory is difficult. Relatively high variances are often associated with estimates of algal productivity in streams (e.g. Bott *et al.*, 1985; Crossey & LaPoint, 1988). The high variation sometimes documented for metabolic measures results in part from spatial variability in environmental parameters (e.g. current velocity) as well as methodological limitations (e.g. differences in current velocity among replicate metabolic chambers). Algal metabolism is also sensitive to environmental variation unrelated to anthropogenic impacts, such as light levels and the timing and frequency of natural disturbances (e.g. spates in streams). Productivity estimates are sometimes less variable relative to structural parameters in experimental streams (Rodgers *et al.*, 1979) and lakes (Morin & Cattaneo, 1992), which are more homogeneous habitats than streambeds. In summarizing findings from a series of laboratory studies, Pratt & Bowers (1992) concluded that, while variability of structural and functional measures was similar, functional parameters were generally more susceptible to changes in resource supply rates, which may confound responses to toxic chemicals and other stressors. Therefore, while functional measures can provide valuable information concerning process rates within

the ecosystem, their reliability as general indicators of ecological impact requires further study.

The use of taxonomic analyses for evaluating ecosystem condition requires a definitive, commonly accepted classification scheme and the availability of qualified personnel for sample processing. Among the algae, diatoms offer several advantages as environmental indicators from the standpoint of measurability. Compared with other common 'algal' groups (e.g. cyanobacteria), the taxonomic classification of diatoms is well understood and widely accepted. Standardized methods for collecting and processing of samples are available in the literature (e.g. see Stevenson & Lowe, 1986). Taxonomic identifications to the species level are based on the characteristics of a ubiquitous structure, the cell wall, and do not rely on observations of specialized structures that are often absent from field samples. Identification aids (e.g. iconographs) and other training and quality control techniques are being established specifically for this group (e.g. Munro *et al.*, 1990; Droop *et al.*, 1993).

The reliability of environmental inferences based on algal taxonomic composition is dependent on the level of taxonomic resolution used when processing samples. Algal identification to the species level where possible is generally recommended since substantial autecological information is lost if coarser identifications are performed (Stoermer, 1978). For example, ecological investigations in North American lakes (e.g. Wolin *et al.*, 1988; Dixit *et al.*, 1989) illustrate extreme variations in environmental optima among diatom and chrysophycean species of the same genus. Coste *et al.* (1991) found an overall high correlation ($r = 0.84$, $n = 1498$) between a species- and genus-level diatom index of water quality in French watersheds, but noted that the correspondence between these two indices decreased markedly with deteriorating water quality conditions. However, considerable resources and expertise are required to perform identifications at the species level, and pollution indices based on algal genera may provide a feasible alternative to species-level indices in some instances (Prygiel & Coste, 1993).

Historically, the analysis of species richness has been an integral part of biological monitoring programs. The accurate estimation of species richness requires both a high level of taxonomic expertise and a considerable investment of time. Based on detailed and laborious counts of stream diatom assemblages, Patrick and co-workers (1954) concluded that several thousand cells must be counted and correctly identified in order to provide an accurate (*i.e.* asymptotic)

estimate of the number of species since most species in these assemblages are present at very low abundances. The use of much shorter counts (e.g., 500 cells) is considered an acceptable means of characterizing assemblage structure (Stevenson & Lowe, 1986). While such procedures provide variable estimates of species richness, these counts do provide valuable information on species evenness and the abundance of dominant species in the assemblage that can be used for evaluating ecosystem condition.

Interpretability

Algal indicators provide ecologically relevant signals of ecosystem change that can be used to distinguish acceptable from unacceptable environmental conditions. Microorganisms, particularly the algae, have been used historically to classify aquatic ecosystems according to the degree of organic enrichment (Kolkwitz & Marsson, 1908; Pantle & Buck, 1955; Fjerdingstad, 1964). The trophic condition of inland surface waters is quantified using algal parameters (e.g. percent bluegreen or nuisance species) that distinguish acceptable from unacceptable states (Hughes & Paulsen, 1990). As for fish and macroinvertebrates, indices of biotic integrity, which score the biological condition of aquatic habitats relative to some reference (*i.e.* minimally-impacted) condition, are under development and have been used to rank streams and rivers according to the extent of environmental deterioration (e.g. Metzmeier, 1991).

Quantification of the condition or 'health' of an ecosystem requires a benchmark or standard for comparison. Historical information on ecological condition is lacking for most groups of aquatic organisms. Paleolimnological analyses of siliceous algal microfossils provides an invaluable database for determining predisturbance ecosystem condition, both biological and chemical, and for reconstructing the rate and extent of past changes in lakes and other aquatic systems with relatively undisturbed sediments (see *Availability of historical database* below). In the absence of reliable historical information, many investigators have adopted the use of regional reference sites for assessing the extent of ecosystem deterioration (Hughes *et al.*, 1986). Ecological conditions in reference, or minimally-impacted, streams are used as a benchmark against which to gauge the condition of other streams in the same physiographic region or 'ecoregion' (*sensu* Omernick, 1987). Algal assemblages, particularly the diatoms, are well suited to this approach for

reasons already discussed (see *Range of applicability* above). In some ecosystems (e.g. lakes and wetlands), the reference site approach may be used in conjunction with paleoecological and other historical methods to improve the characterization of baseline or nominal conditions using diatoms.

Potential for continuity of measurements through time and space

Algal assemblages exhibit excellent continuity through time and with changes in water quality. Although algal species composition undergoes predictable seasonal changes, a species-rich community is maintained throughout the year. Algal assemblages comprise species exhibiting widely differing sensitivities to various environmental stressors. Therefore, while individual populations will not be present under all water quality conditions, the continuity of the community is assured. Thus, consideration of community-level attributes (e.g. taxonomic composition) provides for broader temporal and spatial continuity than reliance on monitoring using specific populations. The broad geographical distributions of many algal species and the ubiquity of these assemblages ensures spatial continuity of these indicators within the context of regional and national monitoring programs.

Appropriateness of the spatial and temporal scale

Because of their small size and consequent sensitivity to environmental changes at small spatial scales, algae can provide important information concerning variation in environmental conditions within an ecosystem (e.g. upstream and downstream of a point source discharge). At the same time, the ubiquitous distribution of algal assemblages in aquatic environments makes them suitable for comparing ecological conditions among ecosystems and for evaluating global changes in the condition of aquatic resources. Whereas highly mobile species (e.g. pelagic fish) integrate the effects of environmental deterioration and relatively large spatial scales, sessile organisms (e.g. attached algae) offer researchers the ability to resolve variation in water quality at finer spatial scales. Benthic algal assemblages are well suited for fine-grained sampling within an ecosystem, whereas phytoplankton assemblages tend to reflect impacts affecting the entire watershed or lake basin. For example, whereas analysis of attached algal assemblages may be useful in tracking the deterioration and recovery of individual bays

within the Laurentian Great Lakes, which are strongly impacted by toxic inputs from specific municipalities (International Joint Commission, 1991), changes in pelagic phytoplankton assemblages reflect basinwide changes induced by nonpoint source nutrient loading into the lower Lakes (Wolin et al., 1988).

Because of the short generation times of algae (e.g. hours or days), individual populations respond rapidly to environmental changes. The applicability of these organisms as early warning indicators is discussed further below (see Anticipatory capability). Algal assemblages provide an integrated expression of environmental impacts over longer time scales (e.g., weeks or months) provided that the influence of normal environmental fluctuations (e.g., seasonal cycles) are accounted for. The historical insight provided by paleolimnological analyses of diatom and chrysophycean cell remnants in sediments provides a continuous record of ecosystem change over decades and centuries that is unparalleled by other aquatic organisms.

Anticipatory capability

The development of an early warning capability for ecosystem monitoring is tied to the identification of environmental indicators that provide a signal of ecosystem deterioration before extensive damage has occurred. Moreover, information contained in these indicators should have a high signal: noise ratio, which reduces the possibility of false signals (Cairns & van der Schalie, 1980), and be capable of being processed and interpreted in a timely fashion.

Because of their short generation times, algae respond rapidly to environmental changes and, thus, may portend impacts on higher organisms, which generally respond on longer time scales. Algae respond directly and immediately to increased nutrient loading, whereas larger, heterotrophic organisms are affected more slowly in response to changes in algal productivity and composition. Numerous studies conducted in the laboratories at Virginia Tech have confirmed that the taxonomic structure and metabolism of naturally-derived protist assemblages are rapidly altered by concentrations of toxic chemicals found to affect the growth and survival of longer-lived organisms (for a summary, see Niederlehner & Cairns, 1994). Field studies to confirm the anticipatory capability of algae and other microbes are noticeably lacking. However, Schindler (1987), in a review of several years of data collected in the Experimental Lakes Area in Canada, concluded that changes in phytoplankton species com-

position and the loss of sensitive species from this assemblage were among the earliest reliable indicators of ecosystem stress observed. Metabolic responses to environmental change, which reflect changes in the physiology of individual cells, precede changes in species composition, which reflect population growth responses resulting from such physiological changes. However, significant changes in taxonomic composition can occur in response to low levels of a stressor in the absence of measurable changes in algal metabolism (e.g. Eaton et al., 1986) due to the high stochastic variability sometimes associated with metabolic indicators.

While the anticipatory capability of microbes is significant, limitations must also be recognized. For example, chronic toxicity in higher organisms can occur in response to low environmental concentrations of chemical contaminants as a result of persistent bioaccumulation in vital organs; models used to explain such effects may not apply to algal cells. The trophic position of algae also limits their ability to accurately predict the impact of chemicals that biomagnify (e.g. pesticides). Thus, false negative signals, that is the indication that ecosystem conditions are acceptable when, in fact, significant environmental damage is occurring, may limit the utility of algal indicators in cases where chronic low level toxicity and bioaccumulation and/or biomagnification processes are involved.

A second problem encountered with early warning indicators is the occurrence of false positive signals, i.e. indications that environmental damage is occurring when, in fact, ecosystem parameters are within an acceptable operating range. The same attributes that allow algal indicators to respond rapidly to anthropogenic stressors also makes them susceptible to fluctuations in environmental condition unrelated to human impact. Measures chosen for use as early warning indicators must exhibit sufficient insensitivity to normal environmental change to reduce the probability of false positive signals to an acceptable level. Indicators exhibiting excessive baseline variability are unsuitable for use as early warning indicators. While metabolic measurements (e.g. algal productivity) often respond rapidly to deteriorations in water quality, these measures are also quite sensitive to extraneous environmental changes (e.g. irradiance) such that the signal:noise ratio may be rather low. As noted above, changes in population and community structure, which result from such metabolic shifts, may provide

more reliable early warning indicators of environmental damage (also see Stoermer, 1984).

Timeliness of information

Early warning indicators of ecosystem change must not only respond rapidly but must also possess information that can be accessed quickly and converted into a form suitable for management actions. While we are unaware of documented comparisons of collecting and processing times for samples of algae and other aquatic groups, it is our experience that algae provide a relatively expedient means of assessing ecosystem condition. The algal assemblage can be exhaustively collected in a much shorter time than for fish and, even, macroinvertebrates. Seining and electroshocking are both time consuming and labor intensive and, in large ecosystems (e.g. deep lakes), may not provide a representative sample of the fish community. Standard protocols for sampling macroinvertebrate assemblages (e.g. Plafkin et al., 1989) are sensitive to differences in substrate conditions among sampling sites and ignore the hyporheic zone, which is an important refugium for many taxa (Williams, 1984) but comparatively difficult to sample. Benthic algae are sensitive to gross differences in substrate type (e.g., plant vs rock), although common taxa can be found in most habitats within the ecosystem.

Laboratory processing times for algal samples should be comparable to that for macroinvertebrates but somewhat longer than for fish. Although techniques for the automated identification of diatom taxa have been attempted (e.g. Cairns et al., 1977; Balfoort et al., 1992), algal enumerations generally must be performed manually and require substantial expenditure of time by personnel trained in taxonomic identification. Although exhaustive counting of algal samples can be extremely time consuming, this is rarely required to obtain reliable estimates of the abundance of dominant species for use in structural indices (see Measurability). Published comparisons of the time required to collect and process samples of different groups of aquatic organisms are needed to further establish the timeliness of algal indicators.

Minimal redundancy with other measures

Algae respond to many of the same environmental stressors as other aquatic organisms, although the relative sensitivity can vary greatly. Algae are direct environmental receptors of increased nutrient loading

into aquatic systems from municipal and agricultural sources and provide sensitive and reliable indications of trophic condition. Herbicides, a second class of common nonpoint source pollutants in agricultural watersheds, also target the algal assemblage because of their phytotoxic properties (Kosinski, 1984). Other commonly used groups of indicator organisms tend to be more susceptible to alterations in the physical habitat, particularly in flowing waters, than to chemical runoff of the types just described. Increased stream temperature and sediment load and water management activities (e.g. channelization, damming) associated with human activity in and adjacent to streams directly affect invertebrate and fish population dynamics (e.g. Cross & Moss, 1987). The taxonomic and functional composition of macroinvertebrate assemblages in particular is closely related to the quality and quantity of allochthonous inputs of organic material (Cummins & Klug, 1979), both of which are affected by alterations in the riparian habitat and the stream channel (e.g. removal of debris dams). Because they tend to be sensitive to different types of environmental changes, algae and aquatic animals provide complementary information regarding ecosystem condition.

Integrative

Given that time and cost considerations place strict limits on the number of environmental parameters that can be measured by monitoring activities, it is clearly desirable that indicators chosen for measurement be capable of summarizing information from many other candidate indicators that cannot be feasibly measured. Strong arguments have been made in favor of the integrative ability of top consumers in aquatic foodwebs, which are usually fish (Ryder & Edwards, 1985). Large, long-lived organisms integrate changes in ecosystem condition over long time scales (e.g. years) and are sensitive to changes in lower trophic levels. Fish tend to be sensitive to physical changes in the aquatic environment as well (e.g. temperature, spawning habitat, refugia, etc.). Fish assemblages have, thus, been favored candidates for providing integrated measures of ecosystem condition (Karr et al., 1986).

Because of their short life span and generation times, algae are incapable of integrating environmental information in the same way as fish and other top predators. However, changes in the algal assemblage do integrate shifts in biomass and feeding efficiency at higher trophic levels and the effects of fluctuations in nutrient availability (e.g. Carpenter et al.,

1985). Furthermore, because they account for a substantial amount of aquatic productivity, algae provide important information regarding energy relationships in aquatic ecosystems. Because of their integral role in ecosystem energetics and biogeochemical cycling, algae provide a relatively unique composite picture of ecosystem condition.

Diagnostic capability

Many measures that are useful for quantifying changes in ecosystem condition (e.g. taxonomic similarity between impacted and reference sites) provide little information regarding the specific cause(s) of deteriorating biological conditions at a particular location. Identifying the causes of environmental impact requires adequate information on tolerances of species to specific anthropogenic stressors. Insufficient and/or unreliable autecological information has hampered the search for suitable indicator species in all taxonomic groups. Because different groups of organisms tend to be sensitive to different types of environmental impacts (see *Redundancy* above), it is unlikely that a single taxonomic group will provide superior diagnostic information in all cases.

Ideally, diagnostic information should be derived using controlled experimentation under environmentally-realistic conditions (e.g. microcosm or mesocosm tests) that is subsequently verified in natural ecosystems. However, as noted by Cox (1992), experimental data regarding species tolerances are generally lacking for the algae. In the absence of rigorous experimental evidence, observational (*i.e.* correlative) information has been used to derive autecological indices for determining the nature of environmental changes associated with ecosystem deterioration. While a sole reliance on correlative information for establishing autecological profiles is problematic because causal relationships can only be implied and not proven, techniques of this type have been used successfully in instances where one or a few stressors are responsible for most ecological effects. The use of algal species as indicators of trophic status has been refined and applied successfully for many years (e.g. Kolkwitz & Marsson, 1908; Butcher, 1947; Pantel & Buck, 1955; Fjerdingstad, 1964; Sládeĕk, 1973; Agbeti & Dickman, 1989). Similar progress has been made in relating historical changes in diatom and chrysophycean assemblages to shifts in lakewater pH (Battarbee *et al.*, 1986; Dixit *et al.*, 1989). Modern multivariate techniques provide an objective means

for classifying species according to their tolerance to major environmental gradients (e.g. Agbeti, 1992). Tolerance data can be used to develop autecological indices for inferring specific changes in environmental conditions.

Successes in developing autecological indices have largely come from work performed in aquatic systems that are strongly impacted by one or few related stressors (e.g. organic enrichment, pH, and heavy metal impacts). However, ecological impacts frequently reflect the cumulative and interactive effects of multiple stressors, both chemical and otherwise. The development of reliable diagnostic procedures thus requires that (1) sufficient autecological information be obtained with which to classify species according to their tolerance to a variety of chemical and nonchemical stressors and (2) the influence of stressor interactions (*i.e.* synergism, antagonism) on species tolerances be understood. Exposure to one stressor can alter algal and microbial responses to additional impacts (Niederlehner & Cairns, 1993). Increased nutrient availability can affect algal responses to toxic inputs (Stoermer *et al.*, 1980; Chen, 1989). Studies such as these suggest that relating changes in biological indicators to individual stressors under conditions of cumulative impact may be difficult. Evaluations of the performance of autecological indices in ecosystems subjected to multiple stressors are needed in order to assess the robustness of these indices for predicting the causes of environmental impacts.

Gaps in the knowledge of environmental tolerances to large numbers of anthropogenic stressors limit field diagnosis of the causes of environmental impact based on biological indicators. Given the problems just discussed (e.g., synergistic interactions among stressors), controlled field and laboratory experiments will undoubtedly continue to play an important role in elucidating the cause of ecosystem change. Details regarding the use and design of such studies are discussed in general terms by LaPoint *et al.* (1989) and for algae and all protists by Cairns *et al.* (1994).

Cost effectiveness

The total cost of collecting, processing, and analyzing samples of stream invertebrate and fish communities for monitoring purposes has been estimated at US$824 and US$740, respectively (Karr, 1991). While we are unaware of similar estimates for algae, the costs should be comparable, if not less. Algal samples can be collected rapidly in the field for taxonomic pur-

poses. Functional parameters such as productivity and production require more intensive field work and/or repeated collection trips and thus, may not be suitable for monitoring programs of limited scope. Laboratory processing of algal samples (*i.e.* mount preparation and a standard 500 cell count) is typically performed at a contract price of about US$50; the time involved for trained personnel to perform a count is similar to that for sorting, counting, and identifying macroinvertebrates. Identification for fish are often less time consuming than for algae (Karr *et al.*, 1986). However, the information content of algal assemblages is high because of the large number of species encountered compared to that for ichthyofauna. When made available, quantitative comparisons of the cost effectiveness of algal indicators should prove favorable.

Availability of a historical database

A major impediment to gauging ecosystem condition is the lack of information concerning the predisturbance condition. The availability of such a benchmark is especially crucial for evaluating the success of ecosystem remediation and restoration efforts (Cairns, 1983). Historical records of biological and water quality conditions are either lacking or far from complete for most ecosystems, and reference ecosystems are often not available for characterizing the minimally impacted condition. Siliceous algae provide one of the few means for reconstructing historical trends in water quality in lakes and other lentic habitats. Stratigraphic profiles of diatom and chrysophycean microfossils in sediment cores provide a historical account of changes in major water quality parameters (e.g. trophic status, pH) in North American lakes dating as far back as the time of European settlement and beyond (e.g. Wolin *et al.*, 1988; Kingston *et al.*, 1990; Stoermer *et al.*, 1990). Paleolimnological analyses of this type have found use in reconstructing historical relationships between environmental deterioration and changes in the condition of economically-valued aquatic resources such as fisheries (Kingston *et al.*, 1992).

Unfortunately, the paleolimnological approach is not easily applied to streams and rivers where sediments are periodically disrupted and resuspended by flooding or other disturbance events (e.g. dredging). In such cases, partial reconstructions of historical conditions may be achieved using algal collections archived in museums and other herbaria. Diatom collections are particularly suited to this type of analysis because of

the ease with which permanent mounts can be made and the number of phycologists who have historically studied this group. For example, diatom collections housed at the Academy of Natural Science in Philadelphia, USA, provide a historical record of biological changes in several major watersheds in North America and elsewhere (e.g. the Amazon) that predate many landmark environmental mandates (e.g. the USA Clean Water Act). Diatom samples recovered from aquatic macrophyte collections made between 1934 to 1958 were used to describe the reference condition of aquatic ecosystems in the Netherlands and to gauge the effects of eutrophication on these waters (van Dam & Mertens, 1993).

Destructiveness of sampling protocols

Sampling of algal assemblages rarely, if ever, results in perceptible environmental impact. Sampling of higher organisms may, in some instances, impact indigenous populations, particularly those of rare or threatened species (e.g. fish electroshocking and associated habitat disturbance in streams). Endangered species currently listed by the United States Fish and Wildlife Service include no algal or other microbial species. Animal rights organizations similarly exhibit little opposition to the removal, testing, or destruction of algae.

Summary and recommendations

It should not be surprising that the organisms often selected for environmental monitoring are those with which regulators and the general public are most familiar. The inclusion of different types of biological indicators in monitoring efforts is often resisted by water managers despite the fact that no one group of organisms is always best suited for detecting the diversity of environmental perturbations associated with human activities. The need for monitoring the status of different taxonomic groups is especially important if the emphasis of environmental management is to shift from the protection of selected ecosystem attributes (e.g. swimmable waters, sustainable fisheries) to the broader goal of ecosystem integrity, *i.e.* the maintenance of ecosystems in robust condition with optimal function.

Algae possess many desirable attributes as indicators of ecosystem integrity and environmental change, most notably the following:

- algae are a ubiquitous and ecological important group in most aquatic ecosystems;
- algae are sensitive to a broad range of environmental stressors;
- algae provide relatively unique information regarding ecosystem condition compared with animal indicators;
- algae respond rapidly to changes in environmental conditions;
- the use of algal assemblages facilitates establishment of a historical benchmark or other reference point for estimating predisturbance condition;
- algae provide a cost-effective monitoring tool in terms of information gained per unit effort.

Numerous approaches exist for using algae as indicators of ecosystem condition as illustrated by recent publications on the topic (e.g. Whitton et al., 1991). While many different approaches have merit, it is our contention that the lack of commonly accepted, standardized protocols for monitoring with algal assemblages has limited the use of this group by regulatory agencies. Although the objectives of individual monitoring programs and, thus, the types of indicators required, will vary, it should be feasible to develop an array of commonly accepted algal indicators and methods for routine monitoring as has been done for other aquatic organisms (e.g. Plafkin et al., 1989). These methods must be standardized to meet the strict quality assurance/quality control requirements of many agencies and to enhance the comparability of information among monitoring programs and geographical regions.

In general, we recommend the use of structural measures of algal condition rather than functional measures, which often require time-consuming field work and tend to be more susceptible to background environmental fluctuations unrelated to anthropogenic impact. And, while taxonomic indicators (e.g. community composition, % sensitive or tolerant species) are more costly to obtain, these measures provide a more reliable estimate of ecosystem condition than the sole use of simple measures such as biomass. This is particularly true in streams where periodic flooding and the sloughing of periphyton can result in dramatic fluctuations in algal standing crop.

Taxonomic indicators based on the analysis of the diatom assemblage appear best suited for routine monitoring. Diatoms are a dominant and ecologically important group in the plankton and benthos of most freshwater and marine ecosystems. Field collection of these assemblages is straightforward and processing methods and taxonomic schemes are widely accepted. In the absence of reference sites, information concerning the tolerance of many diatom taxa to some common types of pollution is available in the literature. Methods are available for gathering historical information on diatom assemblages in lentic habitats.

Both population- and community-based indices have inherent strengths and limitations, and information from both should be used when assessing ecosystem condition. For example, while changes in community similarity can be used to guage the extent of biological changes from the reference condition, analysis of shifts in specific indicator taxa may provide insight into the causes of such change. Continued development of autecological databases is required to increase the utility of population-based indices for detecting impacts associated with different types of pollution. Information currently available on species tolerances to enrichment, acidification, and salinization must be expanded to include responses to other types stressors (e.g. toxic xenobiotics). A key assumption of many population-based indices (e.g. proportion of tolerant or intolerant species [e.g. Lange-Bertalot, 1979]) is that the species most tolerant of one form of pollution will tend to be most tolerant of other forms of pollution as well. This assumption has yet to be evaluated in a falsifiable manner but is certainly amenable to experimental testing in the laboratory and field. Furthermore, most information on species tolerances to specific stressors (e.g. acidification) has been derived from ecosystems where other anthropogenic gradients are absent. The robustness of indices derived from these data need to be tested in ecosystems subjected to cumulative impacts where individual stressor gradients are not so clearly distinguished. Regional calibrations may be necessary for autecological indices, although available evidence indicates that algal metrics transfer reasonably well among geographical areas, e.g. the use of species tolerance information derived in Europe by water managers in the United States.

A variety of functional measures and other biomarkers (e.g. morphological and chemical analyses of dominant populations) have been proposed as sensitive, rapidly responding indicators of environmental stress. While functional indicators have been widely used in laboratory testing, their utility as routine monitoring tools in the field remains to be proven. While metabolic measurements are unquestionably sensitive to changes in ecosystem condition and, logically, should respond more rapidly than changes in cell density or species composition, field studies are

required to show that they provide a consistent, reliable signal of environmental impact without excessive variability or sensitivity to extraneous environmental fluctuations. Alternatively, changes in the abundance of key indicator populations or changes in the growth rate of test populations incubated *in situ* (e.g. Munawar & Munawar, 1987) may provide a more reliable early warning signal of impact.

It is critical that water managers realize that the predictive and diagnostic potentials of any group of indicator organisms are limited. Algae provide reliable and interpretable indications of specific changes in water quality, particularly with regard to trophic status, whereas invertebrate and fish assemblages may better reflect the impact of changes in the physical habitat in addition to certain chemical changes. Because no single group always provides a superior signal of ecosystem changes, indicators derived from several groups should be included when assessing ecological integrity and the success of current and future water management strategies aimed at mitigating complex, cumulative environmental impacts.

Acknowledgments

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