Review

ECOLOGICAL ASSESSMENTS WITH ALGAE: A REVIEW AND SYNTHESIS[1]

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Algae have been used for a century in environmental assessments of water bodies and are now used in countries around the world. This review synthesizes recent advances in the field around a framework for environmental assessment and management that can guide design of assessments, applications of phycology in assessments, and refinements of those applications to better support management decisions. Algae are critical parts of aquatic ecosystems that power food webs and biogeochemical cycling. Algae are also major sources of problems that threaten many ecosystems goods and services when abundances of nuisance and toxic taxa are high. Thus, algae can be used to indicate ecosystem goods and services, which complements how algal indicators are also used to assess levels of contaminants and habitat alterations (stressors). Understanding environmental managers' use of algal ecology, taxonomy, and physiology can guide our research and improve its application. Environmental assessments involve characterizing ecological condition and diagnosing causes and threats to ecosystems goods and services. Recent advances in characterizing condition include site-specific models that account for natural variability among habitats to better estimate effects of humans. Relationships between algal assemblages and stressors caused by humans help diagnose stressors and establish targets for protection and restoration. Many algal responses to stressors have thresholds that are particularly important for developing stakeholder consensus for stressor management targets. Future research on the regional-scale resilience of algal assemblages, the ecosystem goods and services they provide, and methods for monitoring and forecasting change will improve water resource management.

Key index words: biological condition; biomonitoring; diatoms; environmental monitoring; harmful algal blooms; metrics; nuisance algae

Abbreviations: CART, classification and regression tree; CHANS, coupled human and natural systems; EAM, ecological assessment and management; EGS, ecosystem goods and services; LDCI, lake diatom condition index; MMI, multimetric index; SRR, stressor–response relationship

The species composition and biomass of algae, as well as many other characteristics of algal assemblages, are evaluated in assessments of aquatic ecosystems to determine threats to drinking water, fisheries, and recreational uses of water resources (Stevenson et al. 2010). Relating harmful algal blooms in coastal waters to nutrient loads, using paleolimnology to document climate change in arctic waters, and quantifying changes in lake diatom species composition to human disturbance (Anderson et al. 2002, Smol and Douglas 2007, Stevenson et al. 2013) have substantially different goals, but they also represent assessments, broadly defined, of algae in aquatic ecosystems. Thus, characterizations of algal species composition, biomass, metabolism, and chemical byproducts provide assessments of the problems caused by algae, ecosystem services supported by algae, and other changes in aquatic resource conditions that are difficult to measure without algal proxies.

Algae have a long history of use in ecological assessments. Scientific documentation of work with algal indicators of ecological condition started over a century ago (Kolkwitz and Marsson 1908), but that can be predated by Native Americans using bioluminescence of coastal waters to predict mussel poisoning (Meyer et al. 1928). Early taxonomic and ecological studies of algae often included the ecological conditions in which algae occurred, which enabled characterization of species ecological preferences and their use for assessing water quality (Hustedt 1937, 1938a,b, Butcher 1947, Fjerdingstad 1950, Cholnoky 1953). Species ecological preferences have been compiled and amended over the years to provide lists of algal taxa and environmental preferences (Slàdecek 1973, Lowe 1974, van

Dam et al. 1994, Potapova and Carlisle 2011). Zelinka and Marvan (1961) provided a quantitative method for relating species abundances and environmental preferences in a metric of ecological condition that provided the foundation for assessments of human effects on all types of aquatic resources in locations around the world (Watanabe et al. 1986, Wang et al. 2006, Chessman et al. 2007, Kelly et al. 2009, Kireta et al. 2012).

Contemporary ecological assessments with algae can be defined broadly as the application of algal biology to understand relationships among human and natural determinants of algae and ecosystem services. Thus, assessments determine the causes and consequences of growth, accumulation, and death of nuisance and toxic species of marine and freshwater algae for safe use and productivity of drinking water and fisheries (Hallegraeff 1993, Falconer 1999). Assessments use the remains of diatoms and chrysophytes in lake sediments to infer changes in water chemistry and temperature that provide evidence of human contributions to acid rain and climate change (Charles et al. 1990, Smol et al. 2005). Assessments use the species composition of algae in all aquatic habitats to characterize deviations from minimally disturbed condition (Hill et al. 2000, Passy and Bode 2004, Kelly et al. 2008, Stevenson et al. 2013), which has regulatory significance in many countries for protecting water quality. Algal taxonomy, ecology, physiology, and recent advances in genetics and molecular biology are used in these studies. Solving environmental problems requires understanding complex systems, draws from many disciplines of phycology and other sciences, and has many intermediate objectives; but the ultimate goals are relatively focused on the protection and restoration of the final ecosystem goods and services (EGS) that support human well-being (MEA 2005, Boyd 2007).

The goals of this review were to describe the relationships among environmental management and the many types of information generated in algal sciences, thereby increasing the potential for algal research to be integrated and applied in environmental policy. Other reviews have provided detail on current algal metrics being applied in different environmental settings (Porter et al. 2008, Lavoie et al. 2009, Smol and Stoermer 2010, Bellinger and Sigee 2010, Potapova and Carlisle 2011). In this review, I will describe a framework of ecological assessment and management (EAM) as well as how the broad diversity of our research in algal biology can be related to the different EAM processes and policy. I'll cover algal bioassessment broadly to illustrate the commonality in scientific questions addressed across different types of ecological systems and phycological disciplines by using the EAM as an outline for the review. Detailed examples mostly focus on freshwater research, but will reference coastal and marine ecosystems as well.

ECOLOGICAL ASSESSMENT AND MANAGEMENT

EAM can be summarized in four, multistep phases: designing the ecological assessment, characterizing condition, diagnosing causes and threats, and selecting management options (Fig. 1, Stevenson et al. 2004a,b). The first phase, designing ecological assessments, has three basic steps: defining the goals of management and therefore the assessment, developing a conceptual model of how human and natural factors affect valued ecological attributes related to management goals, and determining a sampling plan. Ecological condition of a water body is characterized by measuring current physical, chemical, and biological condition and comparing current observed conditions to expected conditions. Expected condition for ecosystems can be defined a number of different ways, but minimally disturbed condition (Hughes et al. 1986, Stoddard et al. 2006) and desired condition represent two contrasting alternatives that may call for different management strategies (e.g., Stevenson et al. 2004b, Hawkins et al. 2010, Stevenson 2011). Diagnosing causes and threats to management goals involves relating measured current condition, or predicted future condition from forecasting and management models, to contaminants and habitat alterations (stressors) as well as the human activities generating the stressors (Cormier and Suter 2008). Then, the stressors causing or threatening impairment of EGS can be identified by relating observed current condition to predicted effects using stressor–response relationships (SRR) and multiple lines of evidence from field studies, experiments, or modeling (Beyers 1998, Norris et al. 2012). Thus, characterizations of ecological conditions for human and aquatic life uses of waters, stressors, and watershed land use are important for diagnosing causes of current problems, future threats, and identifying
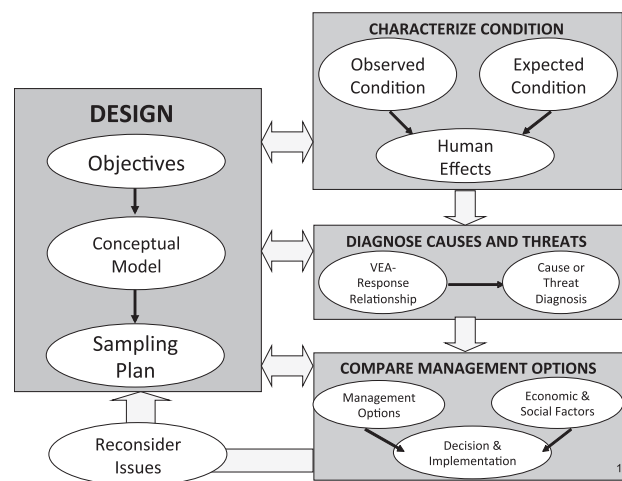

FIG. 1. A framework for assessing and managing ecological systems (Stevenson 2004a, b).

management strategies that can be used to reduce stressors.

Management strategies for ecosystem protection or restoration are selected based on what can be done to reduce stressors, cost–benefit analyses, and socio-political factors that vary with cultures and economies at scales both within and among nations. After management options are selected and implemented, the assessment-management process continues with monitoring of ecological responses to determine if goals are being met and to determine if new problems are developing in response to implemented strategies. At any time during the assessment and management cycle, lack of information may call for returning to the design stage and redesigning the assessment to gather needed information.

DESIGNING ASSESSMENTS: GOALS AND CONCEPTUAL MODELS

Overview. The ultimate goals of assessing ecological systems are to determine status and trends in ecological condition and provide information for protecting and restoring their uses for healthy and safe human communities (USEPA 2011). For practical purposes, whether traditional, institutional, or financial, most assessments have intermediate goals of characterizing attributes of ecological systems that are presumed to provide sustainable support of human well-being. Future assessments will integrate measures of economic conditions and human well-being with those of ecosystems (e.g., MEA 2005). For now, uses for healthy and safe communities, often referred to as "designated uses" in regulatory terms, include safe drinking water, fisheries, recreation, irrigation and industrial use, navigation, and aquatic life use.

Aquatic life use refers to protecting waters for "minimally disturbed conditions," "good ecological status," "protection and propagation of fish, shellfish, and wildlife", and "biological integrity." These phrases for aquatic life use have been codified in the Clean Water Act in the United States (1972, and as amended, U.S. Code 33, Section 101) and the Water Framework Directive of the European Union (European Union 2000), and they have been interpreted by scientists and policymakers into measureable goals, such as indices of biological integrity (Karr 1981, Karr and Dudley 1981) and biological condition (Davies and Jackson 2006). Biological condition is a measure of the similarity in biomass, species composition, and ecosystem function of an assessed site to minimally disturbed or near natural condition. High biological condition that is near nature or minimally disturbed is a state of biological integrity, which is one goal for managing waters of the United States.

Conceptual models provide an overview of how healthy and safe communities are related to specific attributes of ecosystems. They provide a list of variables that could be measured as well as the relationships among variables that will be important for diagnosing causes of problems and developing management plans. Conceptual models should include the ecological system as well as elements of the fully coupled human and natural systems (CHANS) as discussed in MEA (2005), Liu et al. (2007), and Smith et al. (2009). Thus, an overarching conceptual model for integrated assessments of CHANS should emphasize at least four CHANS elements: (i) the physical, chemical, and biological characteristics of the water resources, with specific consideration of ecosystems goods and services; (ii) elements of human well-being; (iii) economic activities; and (iv) stressors (contaminants and habitat alterations) resulting from human activities (Fig. 2). I often add a fifth element, environmental policy, because individual and institutional decision making is an important factor in environmental management (Stevenson 2011). Determining the relationships between algae and the natural and human systems they affect is important for delineating relationships between algae and the ultimate goals for the assessment: designated uses, final ecosystems goods and services, and human well-being.

In this review, I use the phrase "EGS" to refer broadly to structural and functional attributes of ecosystems that directly or indirectly support human well-being. The importance of EGS in ecological assessment is that they serve as final or intermediate endpoints of assessment. They help ecologists focus on what is important in the public debate. Boyd (2007) describes ecological endpoints as purely biophysical, concrete, tangible, measureable, and directly related to human well-being. Boyd and Banzhaf (2007) emphasize distinguishing between intermediate and final EGS, with intermediate services being largely the supporting and regulating services of the MEA (2005), whereas final services are the provisioning and cultural services that have more direct connection with human well-being. In many ways, the designated uses codified in the Clean Water Act can be regarded as final EGS.

Algae in CHANS. Algae are the base of food webs in most aquatic ecosystems, drivers of biogeochemical cycling, and represent significant proportions of biodiversity (Minshall 1978, Wetzel 2001). Protecting natural levels of algal productivity in aquatic ecosystems is important to support food webs and biogeochemical cycling. Maintaining algal biodiversity may be important for sustaining ecosystem function, especially under the threat of regional and global change in environmental conditions (Cardinale et al. 2006, 2012). Maintaining their biodiversity also merits consideration as an ecosystem service for moral and cultural reasons (Lange-Bertalot 1979). Thus, algae generally provide intermediate EGS more than providing final EGS.
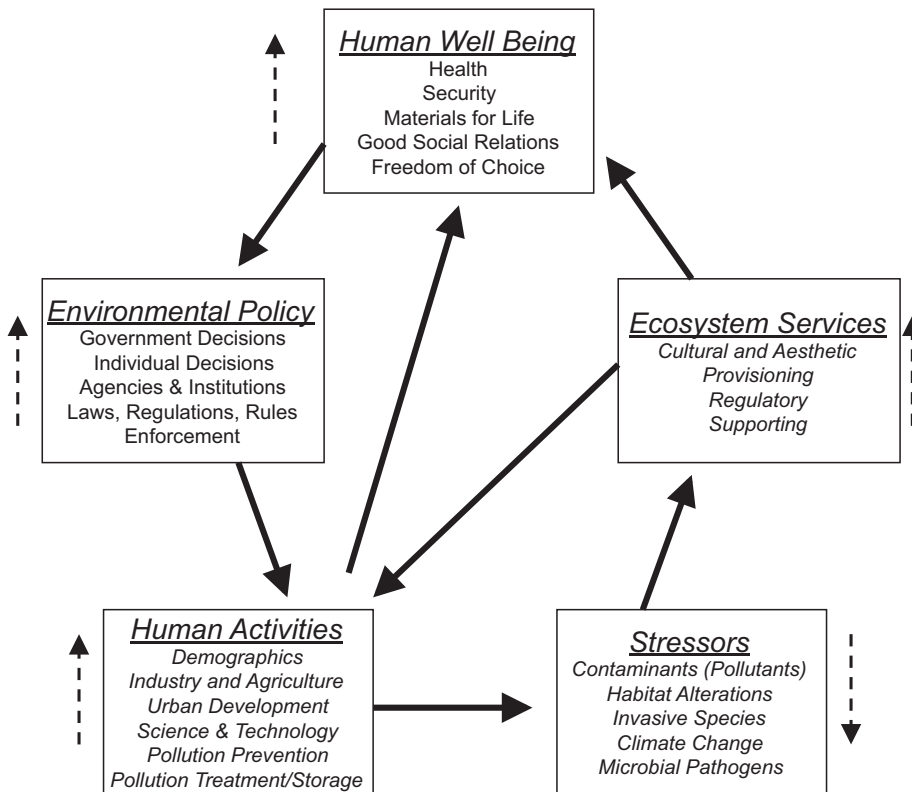
Fig. 2. A conceptual model coupling human and natural systems (CHANS) by linking ecosystem goods and services (EGS) to human well-being and economic activities to stressors. EGS are linked to human well-being directly and indirectly via economic activities within the human activities element. The model includes negative feedbacks on ecosystem services from human activities in the form of stressors. The model also includes feedbacks on the economic activities that produce stressors by individual and institutional decisions about products to buy and regulations to support. The vertical dashed arrows indicate that management goals are high or low levels of conditions within a CHANS element, such that if stressors are managed at low levels by good environmental policies, we can have high levels human activities in watersheds, EGS, and human well-being. From Stevenson (2011).

However, algae cause problems with excessive biomass accumulations and toxic species that alter the physical and chemical conditions of aquatic resources and thereby have negative effects on final EGS. Nuisance growths of algae affect aesthetics in rivers, lakes, and coastal zones for recreational uses by fouling beaches and reducing water transparency (Michael et al. 1996, Poor et al. 2001, Suplee et al. 2008). Accumulations of macroalgae threaten recreation by harboring microbial contaminants on beaches (Ishii et al. 2006) and perhaps in streams that are used for recreation. Algae also produce chemicals that are toxic or have unpleasant tastes and odors that affect drinking water, recreation, and production of fish and shellfish in freshwater and coastal ecosystems (Sigworth 1957, Arruda and Fromm 1989, Carmichael 1992, Bowling and Baker 1996, Falconer 1999, Watson 2004). Excessive algae produce precursors for toxic chemicals (e.g., trihalomethanes) that threatened drinking water (Bukaveckas et al. 2007). Excessive growths of algae deplete dissolved oxygen, elevate pH, and alter physical habitats (Lasenby 1975, Dudley et al. 1986, Holomuzki and Short 1988, McCormick et al. 2001, Wetzel 2001, Antón et al. 2011, Stevenson et al. 2012), thereby altering biodiversity and reducing productivity of aquatic invertebrates and fish.

By explicitly relating algal attributes to ecosystems services, values of ecosystem services, and human well-being, we can better relate costs of management strategies to their benefits for human well-being (MEA (Millennium Ecosystem Assessment) 2005, Stevenson 2011). Algae complexly support a set of final EGS that support human well-being directly with aesthetics and moral services and indirectly by provisioning the economy with natural resources that provide water, food, and chemicals for many economic products. The complexity in these relationships is caused by the possibility that modest increases in algal production have positive effects on some EGS and negative effects on others (Stevenson and Esselman 2013). For example, algal productivity is enhanced by fertilization to stimulate fisheries productivity in habitats ranging from reservoirs to aquaculture farms (e.g., Vaux et al. 1995, Cowx et al. 1998). This produces tradeoffs for different management goals and need for a regional management approach with different waters providing different EGS, as discussed later.

Algal EGS, as manifested by algal biomass, species composition, and metabolism, are regulated directly and indirectly by factors that vary among habitats, seasons, and the species themselves (see Sommer et al. 1986, 2012, Biggs 1995, Stevenson 1997). The ecological determinants of algal metabolism and thereby species composition, biomass, and diversity of assemblages are remarkably similar among habitat types, which should enable transferring problem solving approaches from one ecosystem to another. Although my 1997 conceptual model was designed for benthic algae in streams, it can be extrapolated to algae in other aquatic systems (Stevenson 1997).

Ultimately, climate and geology regulate chemical and hydrological regimes in streams, wetlands, and lakes (Vannote et al. 1980, Frissell et al. 1986, Brinson 1993, Biggs 1995, Soranno et al. 2010) as well as land use in watersheds (Ellis and Ramankutty 2009). These same factors regulate more intermediate factors, such as riparian canopy, suspended sediments, and total nutrient loading, as well as water column and flow stability. These intermediate factors regulate predator abundance, the potential for competition, herbivores, disease, allelopathic interactions, and current velocity. Biotic interactions are most important in ecosystems with natural disturbance regimes in which biota can accumulate to sufficiently high densities that interactions among them are important (Riseng et al. 2004). In lakes, cascading trophic interactions regulate algal species composition and accumulation (Carpenter et al. 1985, Raikow et al. 2004). Most of these ultimate and intermediate factors only indirectly affect algal cell function by affecting the light and inorganic nutrient resources that fuel algal metabolism. In addition, many abiotic stressors (pH, salinity, temperature, sheer stress, abrasion, and toxic substances) regulate resource-fueled metabolism, thereby defining the environmental regimes in which species can live and accumulate. Therefore, taxonomic composition, biomass, and resulting ecosystem functions of algae are most directly shaped by these resources, predation, and abiotic stressors. Excess accumulations of algae caused by high resources can physically alter habitat structure, deplete dissolved oxygen, and elevate pH in ways that negatively affect other biota. When algae foul habitats and produce toxins, they also affect drinking water, recreation, and the provisioning of fish and shellfish.

DESIGNING ASSESSMENTS: SAMPLING PLAN

The sampling plan for ecological assessments varies with goals and phase of the assessment. Surveys, experiments, and modeling represent three contrasting approaches used in ecological assessment. Surveys of water bodies that involve one or more sampling events at each site are designed to characterize the condition of water bodies within a defined area. Historically, ecological assessments often targeted water bodies with problems or particularly high quality systems. In the 1980s and 1990s, the United States Environmental Protection Agency (USEPA) argued for randomly selecting water bodies for statistically unbiased characterizations of a region's and the nation's waters (Herlihy et al. 2000). This random selection of water bodies for sampling often involves stratification to provide sufficient numbers of underrepresented types of water bodies (Peck et al. 2013). Surveys are also valuable for metric development and determining relationships among variables in CHANS, but in this case the random selection of sites should be stratified so a relatively even number of sites is observed at high, medium, and low levels of the independent variables, which are commonly stressors or land use in ecological assessments.

Repeated sampling of a site, or a set of sites, is important for evaluating trends in ecological conditions resulting from human disturbance or restoration, or to confirm current status. Paleoecological assessments are temporal surveys using the upside down records of algae deposited in bottoms of lakes and oceans, as well as some wetlands and rivers (Smol and Stoermer 2010). Paleoecological studies of lake acidification were used to distinguish between natural lake successional processes and effects of anthropogenic drivers of lake acidification (Charles et al. 1990). They provide sufficiently long-term records such that long-term and short-term oscillations in lake and ocean conditions can be distinguished and used to evaluate the role of humans in climate change (Fritz et al. 1999, Smol and Douglas 2007, Winter et al. 2012). On shorter timescales, repeated sampling of sites is used to determine success of management and restorations (Smucker et al. 2014). On even shorter timescales, repeated sampling can provide sufficient data to determine whether a site meets management goals (Stevenson et al. 2010). And finally, continuous monitoring of sites is used to evaluate water quality for drinking water and recreational safety, and even to forecast future problems (Watson 2004, Wynne et al. 2013).

Experimental and modeling approaches are also study designs used in algal assessments, primarily to diagnose stressors and model success of management options. Experiments provide confirmation of cause–effect relationships and parameterization of relationships in models. Past laboratory and field experiments are used as the basis for predicting ecological responses, such as the asymptotic response of algal growth rate to P and N concentrations (Droop 1973, Bothwell 1989, Rier and Stevenson 2006). Experiments can also be valuable during ecological restorations, when causal relationships are not clear and uncertainty has high costs. For example, the perplexing loss of calcareous algal mats with phosphorus enrichment in the Everglades was not well understood until experiments were conducted to isolate the negative phosphorus effect on algae (McCormick et al. 2001, Stevenson et al. 2002, Gaiser et al. 2006). Statistical and process-based modeling help evaluate our understanding of ecological systems, diagnose pollutants, forecast success of restoration, and vulnerability of ecosystems to future environmental change (Litchman and Klausmeier 2008, Thomas et al. 2012, Michalak et al. 2013). Modeling has also been used for assessment when probabilistic sampling was not practical, such as extrapolations of results from small-scale assessments to all sites in a region, nation, or the world (Norris et al. 2007, Downing 2010, Riseng et al. 2010).

Developing the sampling plan also requires determining the number of sites and variables to assess based on the conceptual model, needed data analysis, and financial considerations. Variability in indicators is a key consideration when determining how many sites to sample or how many times to sample a site such that assessments can be made with sufficient precision to detect change. This is one reason that metrics based on algal species composition and traits of species are so valuable in assessments, because they tend to be less variable than one-time measures of some metabolically dynamic physical and chemical conditions (Lavoie et al. 2008, Stevenson et al. 2010). Given that assessments are constrained by financial resources, tradeoffs exist between the number of sites and variables that can be measured. A priori knowledge of intra- and inter-site variability in ecological indicators and accounting for that variability when possible is critical for designing effective assessments.

DESIGNING ASSESSMENTS: SELECTING INDICATORS

Indicators are measures, multimetric indices of measures, or models characterizing ecosystems or one of its critical components (Jackson et al. 2000). Indicator selection depends upon the assessment goals, the variables identified in the conceptual model, and qualities of the indicators. Ecological assessments include characterization of physical, chemical, and biological conditions of water bodies. They also include land use in watersheds that indicate human activities generating stressors and measures of geological and climatic attributes that vary naturally among sites. Good indicators change substantially across a gradient of human disturbance and have little spatial or temporal variability (i.e., high signal:noise ratios, Stoddard et al. 2008). Good indicators are ecologically and socially relevant, broadly applicable, cost-effective, previously measured, technically feasible, diagnostic, and complementary (as reviewed in McCormick and Cairns 1994, Stevenson et al. 2004a). In this review, I focus on algal indicators, which include measures of algal biomass, chemistry, toxins, species composition, and function, for which most have standardized field sampling and laboratory methods (Kelly et al. 1998, Lazorchak et al. 1998, Stevenson and Bahls 1999, Charles et al. 2002, Moulton et al. 2002, Kelly et al. 2009, APHA 2012).

The objectives for using algae in ecological assessments can be measurement of algal characteristics that are directly or indirectly related to management goals or to ecological conditions that affect or are affected by algae. For example, measuring algal species composition and biomass to assess nuisance and harmful algal conditions is an assessment of algal characteristics that is directly related to management goals. In contrast, paleolimnological studies of the acidification rates of lakes or documenting climate change are good examples of how algae are used to characterize ecological conditions that affect or are affected by algae. Use of algae to infer total phosphorus (TP) concentrations in streams, lakes, and wetlands is another example of using algae to serve as a proxy for another ecological attribute. In addition, algae as well as macroinvertebrates, fish, and macrophytes are used in bioassessment as indicators of habitat contamination that threaten other designated uses, because changes in their species composition and biomass reflect contaminants that vary temporally and are too diverse to measure (Jackson and Davis 1994). Among all biota, algae are considered particularly precise indicators of ecological change caused by nutrient contamination and agricultural land use (Hering et al. 2006).

Biomass. Algal biomass is an indicator of threats to drinking water quality, aesthetics, recreation, fisheries productivity, and stressors (e.g., DO and pH) that affect biodiversity of invertebrates and fish. Some indicators of algal biomass are measured in the field, whereas others require collecting samples and laboratory analysis. In the field, we visually assess algal biomass and some attributes of taxonomic composition using Secchi depth in lakes, rapid periphyton surveys in streams, and metaphyton and epiphyton attributes in wetlands (Stevenson and Bahls 1999, Wetzel and Likens 2000, Stevenson et al. 2002). Of course, suspended sediment, dissolved organic carbon, and whiting of water from calcium carbonate can reduce Secchi depth transparency and confound indicators of algal biomass. Rapid periphyton surveys in streams, springs, and beaches provide valuable estimates of macroalgal biomass and genus-level identifications (Stevenson et al. 2012). These visual assessments of condition take relatively little field time, and they account for in-habitat spatial variability in ways that can be related well to drivers, such as nutrient concentrations, despite their less rigorous quantification than laboratory assays of chlorophyll.

Chl a is the most commonly used indicator of planktonic and benthic algal biomass, followed by cell densities and biovolumes. Ash-free dry mass is used to assess benthic algal biomass because shading and nitrogen availability affect chlorophyll density in cells and inorganic sediments. Cell densities and biovolumes determined by microscopic analyses are valuable for determining the proportions of biomass in different taxonomic or functional groups of algae. Algal biomass is considered a good indicator of human disturbance because it monotonically responds to resource and toxic stressors; but challenges exist for relating biomass to stressors because it varies greatly with temporal changes in weather, nutrient loading in coastal zones, and algal scouring in streams and rivers. Biomass also varies spatially within-habitats because of light, current, and substratum variability. Problems with spatial and temporal variability can be solved with large samples sizes

(Dodds et al. 1997), visual assessments, and repeated sampling during a season to get long-term characterizations of biomass (Stevenson et al. 2006, 2012).

Remote sensing using imaging systems on satellites, aircraft, and drones is a rapidly developing technology for characterizing algal biomass and general taxonomic composition of planktonic and benthic algae (Lillesand et al. 1983, Olmanson et al. 2008, Stumpf et al. 2012). With remote sensing, the light in spectral bands and band ratios are used in algorithms as modeled indicators of chl *a* or phycocyanin. These approaches have been used in oceans, large lakes, small lakes, rivers, and streams. Satellite images from SeaWifs, MERIS, LandSat, and MODIS, as well as other satellites, have been used with advantages for each related to differences in pixel sizes, spectral bands, spectral band widths, and times between captured images. Historical satellite images offer a wealth of information that can be used to develop more temporally and spatially extensive characterizations of algae than possible with historic and current data from water sampling. Challenges exist in the large-scale application of remote sensing to multiple habitats because water color varies naturally with alkalinity, dissolved organic carbon, and suspended inorganic sediment. But availability of landscape data with soils, wetlands, geology, and climate attributes of watersheds may help account for natural variability in water color to better predict algal attributes with remote sensing images and standardize methods.

Nutrient chemistry and toxins. Determining the relative importance of N and P regulation of algal accumulation in habitats is challenging because nutrients are sequestered in cells and can be removed from the water column by settling planktonic algae and benthic algae. Elemental ratios of C:N:P are commonly used to determine whether N or P most limit algal accumulation, based on the 106:16:1 ratio observed by Redfield (1958). Lake water measures of C:N:P ratios are largely in particulate nutrient fractions, which are composed largely of algae. Periphyton N and P concentrations are used to determine the limiting nutrient in streams. The nutrient in lowest relative supply is usually considered limiting, unless nutrients concentrations are too high to limit algal growth or nutrient ratios vary with successional stage of community development (Humphrey and Stevenson 1992, Downing et al. 2001).

Algal toxins provide measures of potential algal threats to drinking water, recreation, fisheries, and aquatic life use. Presence of toxin-producing algae is an indicator of a threat, but toxin-producing algae often do not produce toxins. Toxin analysis is challenged by the lack of analytical standards and toxicity equivalence factors (Botana et al. 2009). The World Health Organization established a provisional guideline of $1~\mu g \cdot L^{-1}$ for microcystin-LR, a common cyanotoxin, but not for other algal toxins.

Advances in toxin analysis and assessments of their toxicity are important.

Diversity. Algal diversity is a good indicator of human disturbance because diversity is highly relevant for management; but challenges exist because some algal diversity indices are not accurate measures of species richness, and they vary nonmonotonically along gradients of human disturbance (Stevenson and Lowe 1986, Stevenson et al. 2008a, Blanco et al. 2012). Algal diversity is measured with numbers of species in counts or sample scans, which is sometimes called species richness. Some diversity indices include evenness of taxa abundances, such as S/N (number of species divided by number of organisms), Shannon diversity, or Hurlbert's evenness (Shannon 1948, Hurlbert 1971). However, diversity indices should be used cautiously in assessments because they are not consistently related to human disturbance for three basic reasons. First, species richness is poorly estimated because only minute proportions of algae in habitats are examined and many rare taxa are not observed (Patrick et al. 1954, Patrick 1961). Thus, referring to a measure of taxa numbers in most counts or sample scans as species richness and assuming it is a measure of the number of taxa in a sample or habitat is highly suspect. Second, we might expect the number of species observed in counts to be proportional to the number of species in samples, however evenness of species abundances strongly affects the number of species observed in counts (Archibald 1972, Stevenson and Lowe 1986). Third, species richness could vary nonmonotonically along stressor gradients, for example, positive responses at low levels of resource stressors and negative effects at high levels (Stevenson et al. 2008a). Problems with diversity indices could vary with application. For example, the numbers of species observed in surface sediment diatoms decreased with human disturbance in the USEPA's National Lakes Assessment (Stevenson et al. 2013) and nondiatom diversity decreased with nutrient concentrations in Norway (Schneider et al. 2013). Observed species decreases should be related very cautiously to biodiversity loss. Blanco et al. (2012) discourage use of diatom diversity indices after their evaluation.

Restricting measures of diversity to either pollution sensitive or pollution tolerant taxa in counts could overcome widespread problems with diversity metrics. Using data from a survey of 607 streams in the USEPA's Mid-Atlantic Assessment, we determined which diatom taxa had their highest relative abundances in streams with low TP concentrations, and we designated them as low TP taxa (Stevenson et al. 2008a). We also determined which taxa occurred at minimally disturbed sites, and we called them native taxa. We calculated the expected numbers of native or low P taxa as the average numbers of these taxa at minimally disturbed sites. One diver-

not be measured, historical pH conditions in lakes.

Diatom indicators of TP concentrations in lakes, streams, and wetlands can be used to characterize phosphorus availability when chemistry measurements are not possible, as with paleolimnological pH (Dixit et al. 1999), or when concentrations vary greatly and temporally as a result of diurnal variations in microbial metabolism and weather-related nutrient loading (Stevenson and Smol 2002). If we assume that taxonomic composition of assemblages is regulated by environmental factors over periods ranging from weeks to months, diatom indicators of stressor conditions should provide a "pseudoaverage" of conditions over the time the assemblage developed. Stevenson and Smol (2002) reviewed data and showed temporal variation in measured TP concentration was greater than diatom-inferred TP concentrations in streams and wetlands. Lavoie et al. (2008) showed the integration period to be 2–5 weeks for nutrient indicators in rivers and that the period varied with trophic status of rivers. We have also found that algal biomass in streams across ecoregions of Kentucky, Indiana, and Michigan (USA) is better related to diatom-inferred TP concentrations than average TP concentrations determined over the 8-week period preceding biomass and diatom sample collections (Stevenson et al. 2010). It is possible that variation in bioavailable fractions of TP among watersheds with differing wetland-related DOC concentrations may explain better biomass relationships with diatom-inferred than measured TP. Thus, diatom-inferred TP may be more precise and accurate indicators of phosphorus availability than measured TP concentration, where higher precision is defined as lower variability and higher accuracy is defined as less bias in estimates (i.e., close to the target center). Similar arguments have been made in the Everglades for using diatom TP indices and periphyton chemistry to indicate TP conditions, rather than the highly variable total and bioavailable P concentrations in the water column (Gaiser et al. 2004, 2006, Hagerthey et al. 2012, La Hée and Gaiser 2012).

Kireta et al. (2012) take this analysis one step further in the conceptual model and show algal can be used to infer land use. They do this in very challenging habitats, large rivers. They also show that diatom-inferred changes in water quality of the Missouri, Ohio, and Mississippi are commonly better related to land use than measured water quality. Their results reinforce the value of diatom indicators as robust measures of spatially and temporally variable water quality conditions. In addition, they advance the use of both phytoplankton and periphyton assessments of large rivers.

Taxonomic metrics of biological condition. A second application of taxonomic metrics is measurement of biological condition. Davies and Jackson (2006) list changes in numbers of taxa that are sensitive and tolerant to human disturbance, nonnative taxa, organism condition, and ecosystem function as attributes of biological condition, an important management objective. Metrics for biological condition are likely better measured with subgroups of algae than all algae to independently distinguish responses of species with specific classes of traits, such as: sensitivity or tolerance to pollutants; characteristically found in minimally disturbed or highly disturbed habitats; and growth forms that indicate ecological functions. These metrics are really just special cases of the first type of metric, but in the second case Θ can only take the values of 1 or 0 (i.e., either having the trait or not).

Functions of algal assemblages measured as metabolic rates are important elements of biological condition. However, photosynthesis, respiration, phosphatase activity, and nutrient uptake are difficult to measure in ecological assessments, because they require incubations in the field and field time is often limited (Hill et al. 1998). In addition, metabolic rates are related to algal biomass, which varies temporally in habitats subject to natural disturbance by storm events. Measuring ecological function is more practical in small-scale surveys when time can be allocated for incubations and temporal variation can be managed by sampling at appropriate times after storms.

Other approaches should be considered and tested for assessing ecosystem function. For example models that infer function based on habitat structure, water chemistry, light, algal species composition, and species traits could be less temporally variable and methodologically sensitive than direct measures of metabolic rates. Species morphological traits, growth forms, and potential toxicity have been used as indicators algal function in food webs, biogeochemistry, and EGS, such as drinking water quality. These taxonomic metrics of ecosystem function could be improved by more rigorous modeling of their relationships with measures of ecological functions, but they now serve as a good starting point for characterizing ecological functions.

Little consideration has been given to how characterization of species traits should differ depending upon whether those traits will be used in metrics measuring biological condition or inferring stressors. I would argue that characterizations of species traits by indicator species analysis are used most appropriately in metrics of biological condition, whereas weighted average optima and trait ranks are used most appropriately in metrics inferring stressor condition (Stevenson 2006). Although a full discussion is beyond the scope of this review, the basic premise of my argument is metrics of biological condition should unambiguously and independently characterize changes in sensitive and tolerant taxa. Therefore, separate metrics should be used for sensitive and tolerant taxa. In addition, characterization of whether taxa are sensitive or tolerant to stressors

is best done by using statistical analyses such as indicator species analysis, that quantify the probability taxa show habitat preferences according to a set of defined criteria.

Another consideration is whether metrics should be expressed as proportions of taxa or individuals in assemblages with specified traits. For example, we could calculate our indicator of sensitive taxa to be the proportion of expected low P taxa that were observed in samples (Stevenson et al. 2008a).

$$(S_{low-P,O})/(S_{low-P,E})$$

or as the proportion of expected low P individuals that were observed in samples

$$\sum (p_{i,low-P,O})/\sum (p_{i,low-P,E})$$

Here: $S_{low-P,O}$ and $S_{low-P,E}$ are the numbers of taxa characterized as low P taxa that were, respectively, observed (O) in a sample versus expected (E) in the sample; and $\sum (p_{i,low-P,O})$ and $\sum (p_{i,low-P,E})$ are the sums of proportions of algal cells or diatom valves in the ith species for all low P species that were, respectively, observed (O) in the sample or expected (E) in the sample. Any trait could be substituted for low-P in these metric calculations. The first metric clearly measures proportions of taxa in assemblages with specific traits and habitat preferences. The second metric measures relative abundances of individuals, and better reflects the proportion of biomass that has specific traits. Changes in proportions of taxa can more directly characterize loss and gain in sensitive or tolerant species. Changes in relative abundances of individuals with specific traits more directly characterize changes in function.

Taxonomic metrics of EGS. Boyd (2007) encourages us to focus on endpoints and relate elements of assessment to endpoints of ecological assessment, such as EGS. We could argue measures showing low algal biomasses in water are indicators of high use support for drinking water and aesthetics, which could be defined as final provisioning and cultural EGS. These same indicators for algal biomass are also metrics of productivity, an intermediate ecosystem service, which can have negative effects on other EGS at high levels. Low percent biovolumes of toxin-producing cyanobacteria, dinoflagellates, and diatoms could indicate high levels for drinking water quality, recreation, shellfish, and fisheries. We could argue that indicators of minimally disturbed biological condition indicate waters that have biological integrity and thereby support uses associated with biological integrity (Jackson and Davis 1994), including sensitive native species biodiversity. Thus, algal indicators can be used in assessment as indicators of final or intermediate EGS, stressors, or human activities generating stressors. Improving use

of algal indicators in ecological assessments will be advanced by relating metrics to the way they will be used in assessments and prioritizing management strategies.

Multimetric indices. Multimetric indices (MMIs), pioneered by James Karr (1981) with a multimetric fish index of biotic integrity, have been used and applied in bioassessment with algae (Hill et al. 2000). MMIs are calculated by averaging the values of more than one metric after ranges of metrics are recalculated to be equal (Barbour et al. 1999, Blocksom 2003). MMIs using diatom metrics have been developed for use in streams, wetlands, and lakes (Fore and Grafe 2002, Wang et al. 2005, 2006, Kane et al. 2009, Stevenson et al. 2013). Wu et al. (2012) developed an MMI for large rivers using chl a, the Saprobic Index (van Dam et al. 1994), a cyanobacteria index (Mischke and Behrendt 2007 as cited in Wu et al. 2012), and three diversity indices. Lacouture et al. (2006) developed an MMI for Chesapeake Bay, that was adjusted for salinity, and included chlorophyll, class level biomass indicators, and a species level indicator. MMIs are widely employed in state and national assessments of aquatic resources in the United States. They are hypothesized to respond more consistently to a wide range of human alterations of ecosystems than individual metrics (Karr 1981, Fore et al. 1994, Lacouture et al. 2006). However, Reavie et al. (2008) found some individual metrics had a higher coefficients of determination with human disturbance gradients in the Great Lakes than an MMI. The weaker relationships between human disturbance gradients and MMIs than single metrics in the Great Lakes could be due to lower covariation among stressors along human disturbance gradients in other habitats than the Great Lakes; but further analysis of variability in single metrics and MMIs is warranted.

As with individual metrics, MMIs should be designed for their application. They can be designed with an emphasis to be a composite index that is more precisely and accurately related to human disturbance than a single metric, or an MMI can be a summary index to communicate changes in many attributes of ecological condition with a single index. Particularly in the latter case, goals of the MMIs should be determined and then guide the design of the MMI. Is the MMI intended for summarizing both stressor and biological condition or just biological condition, as indicated in names like Index of Biological Integrity? Some MMIs use indicators designed for stressor identification that include all taxa observed in samples, i.e., not distinguishing responses of pollution sensitive and tolerant taxa. This approach is probably more suited for optimizing development of a composite index of ecological change versus a summary index of biological condition. Karr (1991) and Stoddard et al. (2008) proposed protocols that ensure selected metrics characterize multiple elements of biological

condition, such as diversity, taxonomic, and functional elements. The greatest challenge in MMI design may be determining how much to include in one MMI, because there is so much that can be learned in an assessment with many metrics over many dimensions of physical, chemical, and biological condition. Maybe one of the next steps will be an MMI of EGS, which would include multiple endpoints of environmental management.

Ecological condition is characterized by answering two questions. What is the natural or expected condition? Have human activities affected the natural or expected condition? Implicit in these questions is the assumption that we have goals for managing ecosystems in their minimally disturbed state or some other definition of expected condition. Ecological assessment is more than describing the state of the system. We need to compare observed condition at a site to goals for its management. Therefore, characterizing condition is the comparison of observed conditions at a site to expected conditions. The focus of my presentation on characterizing condition will be "expected condition," to describe its relationship with management goals, and recent advances in modeling expected condition for individual sites by accounting for natural variation.

Expected conditions can be benchmarks for characterizing condition along gradients of human disturbance, such as the good, fair, and poor classifications used in the USEPA National Lakes Assessment (USEPA 2009). Expected conditions can be criteria in government assessment programs that determine whether uses of water bodies are met and whether regulatory action should be taken. In the United States (US), the Clean Water Act establishes the overarching goals of water management, such as drinking water, recreation, fisheries and wildlife, aquatic life support, navigation, and irrigation. These designated uses of a water body are protected by an antidegradation policy and water quality criteria, which with designated uses are the three components of water quality standards for waters of the United States (USEPA 2002). The antidegradation policy is intended to prevent deterioration of waters that meet uses, but these policies often lack sufficient specificity for enforcement. Therefore, water quality criteria are especially important for determining when uses of water bodies are not met. Water quality criteria can be quantitative, such as a minimum of 5 mg · L^{-1} of dissolved oxygen in streams, or narrative, such as "nutrients shall not cause excessive growths of algae." Narrative criteria can have numeric translations (benchmarks) that provide agencies with quantitative and consistent interpretations of narrative criteria. Narrative criteria with numeric translators

provide flexibility to adapt benchmarks when new information becomes available and without formalities required to change criteria. When uses are not met according to the antidegradation policy or criteria, a water body is designated for restoration. This, in brief, is how the US Clean Water Act has been interpreted and applied by federal and state agencies. Interpretation and implementation of water management laws are remarkably similar in the European Union, China, and many regions around the world.

Expected condition can be defined according to uses with legislated criteria, reference condition, or desired condition (Stevenson et al. 2004b). Expected conditions defined by legislation are already written into law and regulation, such as narrative criteria for nutrients and numeric criteria for dissolved oxygen, which are common for US states. Reference condition is defined by levels of human disturbance, such as minimally disturbed, best available (i.e., least disturbed), and best attainable condition (Stoddard et al. 2006). Desired condition is certainly related to legislated condition and reference condition, but in some cases desired condition for some ecosystem services is not minimally disturbed condition such that tradeoffs exist among management goals (Stevenson 2011).

Reference condition is a common goal for water resource management because low levels of primary production and high levels of biological condition (i.e., biological integrity) support many goals of management. Reference condition can be determined by expert opinion, historical data, paleolimnological reconstruction, ecological surveys, and modeling (Hawkins et al. 2010, Soranno et al. 2011). Paleolimnological reconstruction of historical conditions can be used to determine reference condition in water bodies that accumulate sediments, such as lakes, wetlands, some rivers and estuaries, and oceans (Dixit et al. 1999, Slate and Stevenson 2000, Smol and Stoermer 2010). Diatom frustules, chrysophyte scales and cysts, and pigments in sediments of lakes and wetlands have been used in indicator models to infer historical conditions and assess change through time (Leavitt 1993, Smol 1995). Reference condition is also characterized by assuming that current conditions in minimally disturbed watersheds represent historical natural conditions (Hughes et al. 1986).

Reference condition based on current conditions at sites can be determined using a subset of sites or by using all sites (Hughes et al. 1986, Hughes 1995). Routinely, reference condition is characterized by selecting a subset of sites that meet criteria for minimally disturbed, best available, or best attainable sites (Hughes et al. 1986, Stoddard et al. 2006). Then assessment benchmarks for metrics or MMIs can be established as the 25th percentile of the reference site distribution or another percentile deemed appropriate.

To account for natural variation among reference sites, reference sites can be subdivided by ecoregions in which sites have relatively similar geological and climatic conditions (Hughes et al. 1986, Omernik 1987). Alternatively, site-specific models for characterizing reference condition can be developed to account for natural variation in conditions found within ecoregions (Hawkins et al. 2010). Site-specific models have been developed in which the observed proportion of expected reference algal taxa are assessed for a site (Chessman et al. 1999, Cao et al. 2007), which is expected to decrease with increasing human disturbance. This approach was pioneered with invertebrates (Moss et al. 1987), which is used extensively with invertebrates in Australia, Europe, and the United States. Site-specific models for other algal metrics, such as trophic diatom indices, have also been tested by using a ratio or residuals relating the observed index value and the expected value at the site, with expected index values predicted by regression models that account for natural variation related to alkalinity, sampling season, mean annual air temperature, and precipitation (Cao et al. 2007, Kelly et al. 2009). Natural variation in metric values can be as great as effects of human disturbance (Stevenson et al. 2009).

During the analysis of data for the USEPA (2009) National Lakes Assessment, we compared use of site-specific modeling approaches for predicting expected reference condition of a multimetric Lake Diatom Condition Index (LDCI) with ecoregion and lake typology classifications of reference condition (Stevenson et al. 2013). For the National Lakes Assessment, surface sediment diatom assemblages were determined for 1031 lakes, a set of best metrics was determined, and an MMI was developed. To account for the great natural variability among lakes across the United States, we decided to determine the expected MMI for a site as if the site were in minimally disturbed reference condition, and assess lakes as the deviation in observed LDCI from expected reference LDCI (i.e., the adjusted LDCI). Using this approach, an adjusted LDCI greater than zero indicated the site had condition better than we expected.

To compare methods for calculating expected reference LDCI, we used LDCI at reference lakes only and calculated: (i) average LDCI for lake types defined by ecoregions, natural versus man-made, or by cluster analysis of lake attributes affected little by human disturbance; and (ii) site-specific predictions of expected reference LDCI based on regression models for reference LDCI predicted by geological, climatic, geographical, and soil characteristics of sites and their watersheds. We found that site-specific models for reference condition explained substantially more variation in the expected reference LDCI than any lake typology. In our selected model, expected reference LDCI was negatively related to watershed mean soil erodibility and basin:lake ratio,

and it was positively related to latitude, longitude, elevation, and summer precipitation. These relationships have reasonable ecological underpinnings because we had hypothesized that LDCI would respond strongly to nutrients, as a dominant stressor of algae in lakes because watershed mean soil erodibility and basin:lake ratio would be positively related to nutrient concentration in lakes; and latitude, longitude, elevation, and summer precipitation would be negatively related to nutrients because summer precipitation could dilute nutrients concentrations and reference site quality would increase with latitude, longitude, and elevation. The latter determinants of climate should negatively affect the regional extent of agriculture and row-crops requiring fertilizer, and therefore may reflect regional variation in quality of reference sites.

Site-specific models of expected reference condition were also used in the USEPA's National Rivers and Streams Assessment (USEPA unpublished results). In this case, the models for expected reference condition were calculated with random forest methods, because random forest models performed better than multiple linear regression. In addition, expected reference condition was calculated for individual metrics rather than the MMI. This approach for modeling expected reference condition of metrics or MMIs for each water body and then assessing condition with the deviation between observed and expected condition has been referred to as modeled MMIs. For both the National Lakes Assessment and National Rivers and Streams Assessment, the 25th and 5th percentiles of modeled (aka adjusted) MMI values for reference water bodies were used as benchmarks to delineate assessments of good, fair, and poor condition.

Expected reference condition for a site can also be determined with measures of ecological conditions of all sites, not just reference sites (Hughes 1995). With this approach, models of expected condition for physical, chemical, or biological condition are developed using indicators of human disturbance as well as indicators of natural variation among sites (e.g., geology, climate, etc.). Human land uses in watersheds (e.g., percent agriculture or urban) are commonly used as indicators of human disturbance. Minimally disturbed or best attainable condition is calculated for sites by setting land use indicators to zero or appropriately close to zero in the models. Then deviation in observed condition from expected condition is calculated to assess human alteration of condition at sites. Expected reference models using data from all sites in a region are sometimes called "dirty models," in contrast to models using only minimally disturbed sites, which are called "clean models." Dirty models have most commonly been used to determine minimally disturbed condition for water chemistry (Smith et al. 2003, Dodds and Oakes 2004, Stevenson et al. 2008a), but they have also been used for biological

metrics (Stevenson et al. 2009). Using all sites to determine reference condition may be necessary in ecoregions with few reference sites because watersheds are extensively altered by human activity (e.g., anthropogenic biomes suitable for agriculture) or when habitats naturally vary greatly and each typology has few examples (e.g., wetlands). Dirty models can also be valuable for providing a standardized scale for qualifying reference condition when minimally disturbed and best available conditions are known to vary among regions with varying extents of alteration by human activities.

CHARACTERIZING CONDITION: SRR & CRITERIA

Quantitative relationships among elements in CHANS are important for understanding feedbacks, tradeoffs, and thresholds in these complex systems. One class of these relationships, those between stressors and ecosystem services provide valuable justification for stressor management goals and environmental criteria (Stevenson et al. 2004b, Cormier et al. 2008, Stevenson et al. 2008a, Dodds et al. 2010). These are often called SRRs based on similar analyses in human health, ecotoxicology, and risk assessment. SRRs describe the loss of valued ecological attributes (i.e., EGS) with increasing pollution or habitat alterations. They can be used to diagnose the stressors causing or threatening to cause problems, and they can quantify the costs and benefits of better stressor management.

SRRs are valuable complements to reference approaches for establishing management criteria, particularly stressor criteria, because reference approaches do not explicitly relate changes in EGS to stressors. Establishing criteria by SRRs is referred to as an effects-based approach. An ideal example of criteria development involves three steps (Fig. 3,

Stevenson and Smol 2003, Stevenson et al. 2004b): (i) find an ecosystem service that responds nonlinearly to a stressor; (ii) establish the stressor criterion at a level that protects the ecosystem service; and (iii) establish a complementary biological criterion using a biological indicator or MMI that responds linearly to the stressor and at an indicator or MMI level predicted to occur at the stressor criterion. SRRs that have some assimilative capacity (resilience to low levels of disturbance) are particularly valuable because they delineate a point along the stressor gradient, the threshold, where risk of losing a desirable attribute or gaining an undesirable attribute suddenly increases with steady incremental increases in stressor level (Muradian 2001, Stevenson et al. 2008a). Other types of nonlinear responses, such as positive and negative asymptotic relationships, are too sensitive to stressor change for criteria development because the threshold in response marks the upper stressor levels where the full response has been manifested, and the relative linearity in the response region above that threshold does not provide a distinct stressor level for justifying criteria (Stevenson et al. 2008a). Thresholds are distinguished conceptually from benchmarks (Cormier et al. 2008), where the latter are any point along an environmental gradient that can be justified as a candidate for environmental criteria. In this article, I use "thresholds" to mean benchmarks delineated by nonlinearities in SRRs.

Thresholds can be found in many algal responses to stressors, particularly nutrient concentrations. Asymptotic relationships between stream algal biomass and nutrients in experiments and field surveys (Bothwell 1989, Dodds et al. 1997, Rier and Stevenson 2006) showed thresholds around 10 μg $SRP \cdot L^{-1}$, 30 μg $TP \cdot L^{-1}$, and <100 μg $DIN \cdot L^{-1}$, where SRP is soluble reactive phosphorus and DIN is dissolved inorganic nitrogen. Thresholds in *Cladophora* cover on stream bottoms occurred at 23 and 27 μg $TP \cdot L^{-1}$ in separate studies (Stevenson et al. 2006, 2012). Thresholds in diatom species composition of streams occurred at multiple benchmarks from 10 to 82 μg $TP \cdot L^{-1}$ (Stevenson et al. 2008a, Smucker et al. 2013a). Thresholds in floating calcareous algal mats of the Everglades were observed at 10 μg $TP \cdot L^{-1}$ in field surveys and much higher concentrations in experiments (McCormick and Stevenson 1998, Stevenson et al. 2002). Downing et al. (2001) show thresholds in % cyanobacteria in lake phytoplankton at 10 and 100 μg $TP \cdot L^{-1}$. Soranno et al. (2008) found some evidence for thresholds in chl *a* between 30 and 40 μg $TP \cdot L^{-1}$, which was also a concentration associated with change in phytoplankton species composition. Smucker et al. (2013b) show thresholds in stream algal responses at specific levels of land-use land-cover metrics (impervious cover, riparian buffer, and riparian wetlands in watersheds). Thresholds in algal responses are also likely along other stressor gradients. For example,
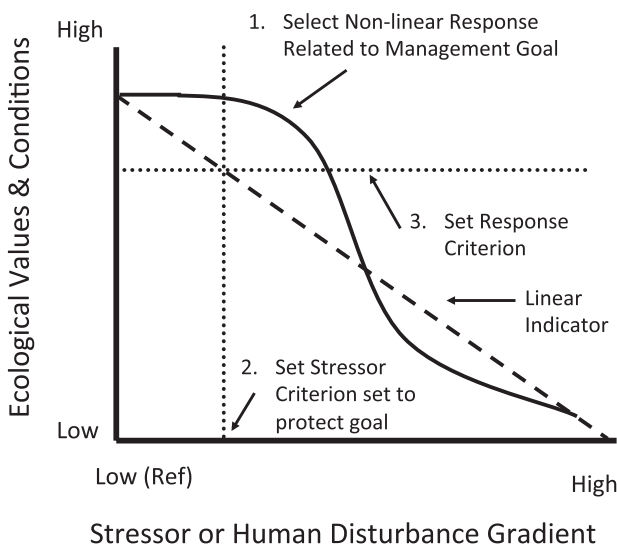

FIG. 3. Steps for using threshold responses to set both stressor and biological criteria (modified from Stevenson et al. 2004b).

R. Pillsbury, R. J. Stevenson, M. D. Munn & I. Waite (unpublished data) observed greater change in stream diatom species composition in the circumneutral range of pH than high or low pH ranges.

Resolving nonlinear responses requires sufficiently high sample size and often repeated sampling of habitats to precisely characterize conditions at a site. In our work characterizing algal and invertebrate responses to nutrient concentrations in streams (Riseng et al. 2004, Stevenson et al. 2006, 2012), we found repeated sampling of streams was necessary to characterize algal biomass and nutrient concentrations because storm flows caused great variation in both variables. We also found that 70 sites enabled observation and statistical confirmation of nonlinear trends.

Statistical confirmation of nonlinear trends is important because some analyses, such as classification and regression tree (CART), which is commonly used to detect thresholds, will delineate a threshold (change-point) in an absolutely linear response (Daily et al. 2012). However, the CART model will explain less variation in the relationship than a linear model in such cases. So, linear and nonlinear models, whether CART or piecewise quantile regression, should be compared to evaluate whether nonlinear responses can be justified and where benchmarks for management should be established (Daily et al. 2012, Qian and Cuffney 2012, Baker and King 2013). In addition, CART models may delineate the upper or lower end of a stressor range in which responses are unusually sensitive; and the lower end of the sensitive range is important for protecting uses (Stevenson et al. 2008a).

Thresholds can be found in many relationships in CHANS and are particularly important for managing ecological systems (Stevenson 2011). Threshold responses can propagate through CHANS to stimulate management decisions. *Cladophora* responses to stream phosphorus enrichment and recreational users' perceptions of *Cladophora* provide a good example of how a small increase in a stressor can cause a proportionally greater change in EGS. Filamentous green algal cover of stream bottoms increases from less than 5% to greater than 20% cover at thresholds of 23 and 27 µg TP · L^{-1} in Kentucky and Oklahoma (Stevenson et al. 2012). Suplee et al. (2008) evaluated recreational users' perception of streams in Montana having different levels of benthic filamentous green algae and found a nonlinear response with user perception decreasing most rapidly, from over 90% to less than 30% desirable response, when algal biomass ranged from 110 to 200 mg chl a · m^2. Thus, constraining filamentous green algae to ≤110 mg chl a · m^2 is important for supporting high recreational uses of these water bodies. According to relationships between % cover of stream bottoms by filamentous green algae and mg chl a · m^2 for Kentucky streams

(Stevenson et al. 2012), a stream has >110 mg chl a · m^2 if bottom cover by filamentous green algae is greater than 7%, and it has >200 mg chl a · m^2 if bottom cover is greater than 18%. Therefore, a threshold response in filamentous green algae between 20 and 30 µg TP · L^{-1} spans the user response threshold from 110 to 200 mg chl a · m^2. As a result, risk of losing recreational value of a stream can increase dramatically with TP increases from 20 to 30 µg TP · L^{-1}, particularly in streams with shallow rocky areas serving as optimal habitat for filamentous green algae. I have referred to the phenomenon when one threshold response triggers a successive threshold response in CHANS as propagating thresholds (Stevenson 2011), which is somewhat like a domino effect. In this case, a threshold response in the stressor–EGS relationship triggers a threshold response in the EGS-human well-being relationship, where TP concentration is the stressor, % filamentous green algal cover is a negative indicator of aesthetics (an EGS), and user perception of the resource is an indicator of human well-being. If the biomass response to TP pollution was too low or was linear, the effects of modest increases in nutrient pollution might not cause public concern. Propagating thresholds increase importance of management response because benefits associated with valuation of ecosystem services are more likely to exceed costs of pollution management when thresholds propagate through CHANS.

Risk analysis is another important consideration for selecting levels of stressors that will adequately protect ecosystems services. Risk analysis calls for defining loss of an attribute and then quantifying the probability that condition will occur. For example, how often is losing an important attribute acceptable? What level of risk is acceptable for losing an attribute? What is the probability of losing the attribute (i.e., the risk) as stressor levels increase? In the above examples where we used regression models to describe an SRR, the models predicted the central tendency of relationships, which means that a valued attribute is greater than the predicted level 50% of the time and less than that level 50% of the time. Absence or loss of a valued attribute 50% of the time is commonly too high. Thus, guidelines for criterion development call for providing a margin of error, which is satisfied by moving the stressor criterion a safe amount lower than the threshold.

Downing et al. (2001) incorporated risk and thresholds in their characterization of cyanobacterial response to TP. The cyanobacterial proportion of phytoplankton and the risk of cyanobacteria being greater than 50% of phytoplankton biovolume were related to TP concentration in Iowa lakes. The central tendency in cyanobacterial proportion of phytoplankton increased gradually over the gradient of TP from <5 to >1000 µg · L^{-1}. Two thresholds were visually evident in the relationships: one at <10 µg

TP · L^{-1}, where cyanobacteria were never >20% of phytoplankton biomass and another near 100 μg TP · L^{-1}, above which cyanobacteria were between 30% and 95% of phytoplankton biomass. Risk of cyanobacteria being greater than 50% of biomass was less than 10% if TP was less than 40 μg · L^{-1} and increased to 80%–100% above 100 μg TP · L^{-1}. Thus, different benchmarks for nutrient criteria existed along the TP gradient with this one algal parameter based on average conditions and a risk analysis.

Geology, climate, landscape position, and hydrogeomorphology are natural determinants of ecological response to human disturbance as well as the expected condition in a minimally disturbed watershed. As an example, stream algal and invertebrate responses to nutrients differ in Kentucky and Michigan ecoregions with differing hydrological variation and summer drought stress (Riseng et al. 2004, Stevenson et al. 2006). Deep glacial tills in Michigan store rainfall, thereby reducing flashiness of storm events and also maintaining modest baseflows in streams during droughts. More severe floods and droughts in Kentucky constrained aquatic invertebrates in Kentucky streams to much lower abundances than in Michigan streams. As a result, benthic diatom abundance responded relatively little to nutrient enrichment in Michigan streams because grazing invertebrates controlled their accumulation. On the other hand, benthic diatoms responded very sensitively to low nutrient enrichment levels in Kentucky and reached much higher masses than in Michigan streams. The between-ecoregion difference in diatom responses to nutrients were greater than filamentous green algal responses, which was likely related to greater grazer control of diatoms than filamentous green algae.

"Tiered uses" is a policy employed by some states in the United States that provides protection for high quality waters and incremental restoration goals for lower quality waters (Davies and Jackson 2006). Tiered uses is based on the concept that biological condition changes predictably along multistressor gradients of human disturbance with successively greater loss of the native species and ecosystem functions characteristic of minimally disturbed waters as stressors increase. Thus, multiple benchmarks for protecting ecosystems are expected to occur along a gradient of stressors or indicators of human disturbance. Those benchmarks can be used to set expectations for different tiers of biological condition for different water bodies. For example, criteria of 10, 40, and 100 μg TP · L^{-1} can be justified for different cyanobacterial threats in temperate zones lakes based on data from Downing et al. (2001). Although 10 μg TP · L^{-1} is a desirable goal for lake management to prevent high cyanobacterial percentages in lakes, restoration of lakes in highly altered watersheds to 10 μg TP · L^{-1} could be impractical; but 40 and 100 μg TP · L^{-1} benchmarks

could provide incremental restoration goals. Stevenson et al. (2008a) proposed the application of tiered uses for streams with TP concentrations <10 μg · L^{-1} protecting sensitive native taxa and <30 μg TP · L^{-1} controlling risk of nuisance growths of filamentous green benthic algae. Thus, the policy of tiered uses allows for different levels of designated uses for different water bodies.

Having multiple benchmarks for protecting ecosystems along a gradient of stressors or indicators of human disturbance also allows for accounting for natural variability when establishing stressor criteria. For example, some lakes naturally have TP concentrations >10 μg · L^{-1} (Soranno et al. 2008). Therefore, the 40 and 100 μg TP · L^{-1} benchmarks from Downing et al. (2001) provide options for TP criteria in lakes with naturally high TP concentrations. Thus, we can develop predictive models for natural concentrations of nutrients for individual lakes based on surrounding geology, climate, landscape position, and their hydrogeomorphology; then lake-specific nutrient criteria can be established for individual lakes based on their predicted minimally disturbed condition and ecological responses to nutrient pollution (Soranno et al. 2008). Similarly different expectations could be established for parts of lakes or coastal zones where nutrient concentrations are naturally higher, such as nearshore waters and embayments near river mouths compared to offshore waters. Different management goals for nearshore and offshore waters or different longitudinal zones in reservoirs may provide more sensitive assessments of response to nutrient changes than monitoring one location with one set of expectations.

Refinement of SRRs becomes even more important in environmental management as policies start to incorporate ecosystem services as management endpoints and to ensure more likely achievement of management goals without overprotection. The responses of ecosystem services of waters are not concordant along human disturbance gradients (Stevenson and Sabater 2010, Stevenson 2011). Tradeoffs exist in uses of waters for different ecosystem services (Fig. 4). This problem increases as we consider the international and regional variations in cultures and economic conditions that affect valuations of EGS. For example, along a nutrient pollution gradient, we can expect a shift from high support of algal and invertebrate biological condition, aesthetics, and drinking water quality at low nutrient concentrations to increases in fisheries production without much threat to the biological condition of fisheries at intermediate levels of nutrient pollution (McQueen et al. 1986, Peterson et al. 1993, Miltner and Rankin 1998). High nutrient concentrations produce few beneficial uses in many parts of the world because the risk increases greatly for nuisance and harmful algal growths, oxygen depletion, and loss of high quality fisheries. But in
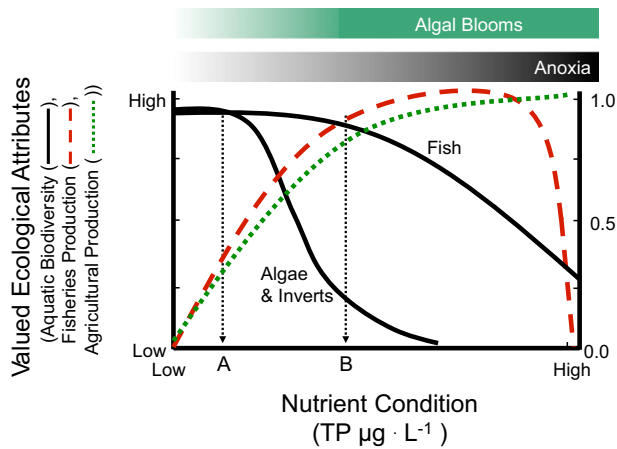
Fig. 4. Tradeoffs among uses of rivers indicated by hypothetical relationships between a resource stressor (e.g., nutrient concentrations) and a suite of ecosystem services of catchments: drinking water impairment by algal blooms; algal, invertebrate, and fish biodiversity; fisheries production; and agricultural production. The vertical lines (A and B) indicate nutrient criteria that could be used to protect different uses in different watersheds, e.g., all aquatic biodiversity at A and fish biodiversity, fish productivity, and agriculture at B (modified from Stevenson and Sabater 2010).

some rural and poor communities, ponds function as human and animal waste disposal sites that produce crops of fish and local waterways are needed to transport wastes downstream. Given that alterations of watersheds for economic activities associated with agriculture and urban development benefit human well-being and those activities generate pollutants that degrade aquatic EGS, real tradeoffs exist for resource managers to balance protection of water resources and allow watershed alteration to support economies. Thus, different uses should be designated for different waters to optimize ecosystem services across a landscape, to accommodate different valuations of EGS with culture and economic status, and to enable a mechanism for structuring international environmental policy (National Academy of Science 2012). A refined understanding of relationships between human uses of watersheds, ecosystem services, and stressors will provide the scientific foundation for improved management of different waters for different uses and optimizing EGS across regional and global landscapes.

DIAGNOSING STRESSORS

Diagnosing stressors in ecological assessment calls for listing possible human alterations of ecosystems that could be causing problems, characterizing likely stressors, and evaluating a wide range of evidence. Diagnosing stressors is also referred to as stressor identification (USEPA 2000), causal pathway analysis (Cormier and Suter 2008), or cause and effect analysis (Norris et al. 2012). Hill (1965) proposed

nine causal criteria: plausibility; coherence; analogy; temporality; strength, consistency, and specificity of association; experiment; and SRRs. The causal criteria provide lines of evidence for identifying the likely stressor or stressors causing a problem. These causal criteria were proposed for human health applications and have been refined for ecotoxicology by numerous authors (Fox 1991, Beyers 1998, USEPA 2000, Norris et al. 2012). Causal criteria should be considered when designing assessments by including variables in the conceptual model that are plausible stressors with conceptually sensible reasons for causing the problem. Coherence is complementary to plausibility and means that the cause–effect relationship is consistent with known information. Analogy refers to similar stressors causing similar problems. Temporality refers to timing with cause occurring before effect. Strength, consistency, and specificity of the association call for evidence that the problem has a high probability of occurring if exposed to the stressor, that stressor and problem co-occur in other ecosystems, and that no other stressors cause this problem. Experiments are important for establishing that a stressor or interaction among multiple stressors can cause a specified effect. SRRs show that incremental increases in stressors correspond to magnitude of the effect, and they provide the basis for developing stressor criteria.

If we assume that we have gathered information on all plausible stressors, including the correct stressor, then stressor diagnosis should be a matter of comparing observed condition to predicted responses of the ecosystem based on SRRs to determine which stressor is sufficiently great to cause the problem. Erroneous conclusions can result from this approach if multiple stressors covary with the causal stressor and we have incomplete information on SRRs of individual stressors. Multiple interacting stressors can confound SRRs. Experimental confirmation of cause–effect relationships and process-based modeling can be important complements to SRRs in stressor diagnosis.

The loss of floating calcareous algal mats in the Everglades provides a good example of stressor diagnosis, criteria development, and results of algal assessments being used in management. The Everglades is a vast mosaic of wetland types in south Florida, but is predominantly vegetated slough and sawgrass marsh. Water in the Everglades flows from the north to the south, originating from Lake Okeechobee and nearby agricultural lands, where it ultimately discharges into Florida Bay (Davis and Ogden 1994). Because of flood concerns, the hydrology of the Everglades is managed via a network of drainage canals, dikes, and water control structures (gates and pump stations), which allow canal water into marshes. Three major problems were visually evident in Everglades marshes that received anthropogenically enriched canal water:

species composition of the dominant aquatic plants changed from sawgrass (*Cladium*) to cattail (*Typha*); aquatic macrophytes often filled in the normally open water sloughs; and calcareous algal mats were eliminated (McCormick et al. 2009).

Florida's narrative nutrient criterion states *in no case shall nutrient concentrations of a body of water be altered so as to cause an imbalance in natural populations of aquatic flora or fauna*. The Everglades Forever Act of 1994, legislation passed by the Florida State Legislature, reaffirmed this criterion and called for numeric nutrient criteria for phosphorus. The changes in macrophytes, slough habitat, and floating calcareous algal mats in Everglades marshes that received anthropogenically enriched canal water were considered "imbalances in flora or fauna", and violations of Florida's narrative nutrient criteria. To develop a numeric TP criterion to serve as a translator for narrative criterion, algal assessments were used to diagnose stressors (i.e., demonstrate that TP was a causative pollutant), to characterize responses to TP, and to guide legally defensible management decisions.

Calcareous algal mats are a characteristic feature of low TP, karstic regions of the Everglades, and are important sources of primary production and calcareous sediments (Browder et al. 1994). Floating calcareous algal mats are one of the most evident accumulations of algae in the Everglades. While they often start as periphyton on macrophytes, they can grow into floating mats that cover the surface of open water areas and can appear to be bleached white. These mats are composed of cyanobacteria, diatoms, bacteria, and precipitated calcium carbonate. Loss of floating calcareous algal mats, as well as significant reductions in the prevalence of sensitive diatoms, was one of the key problems attributed to discharge of canal water into marshes. Many researchers hypothesized that phosphorus was the important contaminant in canal water that was causing loss of the floating calcareous algae. Plans for Everglades restoration required that the causative pollutant be clearly identified before billions of dollars in public and private restoration funds were allocated to reduce phosphorus contamination. A simple conceptual model for algal ecology in the Everglades would include natural determinants: light and nutrients, pH to regulate algal species composition and calcium carbonate precipitation in particular to support characteristic periphyton development, grazers, physical disturbance from animals, rain, and wind. Canal waters were expected to change water depth and carry many contaminants, including nutrients and potentially toxic agrochemicals. Early marsh surveys and diatom paleoecology documented that nutrient concentrations were very low prior to human disturbance in the area, and that nutrients probably limited algal production (Flora et al. 1988, Vymazal et al. 1994, Slate and Stevenson 2000). Canal waters tended to be higher in

pH, which would not explain shifts in species composition and loss of calcium carbonate. Therefore, it was hypothesized that adding phosphorus, probably the most important nutrient limiting growth rates of algae, caused the disappearance of calcareous algae from otherwise minimally disturbed aquatic habitats of the Everglades.

Short-term and long-term experiments in mesocosms in the Everglades confirmed that manipulations of phosphorus alone caused loss of the calcareous algal mats and other changes in algal species (Craft et al. 1995, McCormick and O'Dell 1996, McCormick and Stevenson 1998, Pan et al. 2000). Additional experiments showed that adding phosphorus increased algal growth rates, but phosphorus reduced calcium carbonate deposition in mats and reduced the ability of floating mats to withstand wind and rain disturbance when floating on the water surface.

The next question was, "As a protection and restoration target, what phosphorus level will allow for the continued growth and maintenance of the calcareous algal mats?" Results of experiments could be used to establish criteria if they sufficiently simulated long-term marsh responses to phosphorus enrichment. Another approach was using SRRs between floating calcareous algal mats and TP along nutrient gradients generated by canal water releases into marshes (Pan et al. 2000, Stevenson et al. 2002, Gaiser et al. 2006). We used aerial photography from a helicopter to determine the percent coverage of sloughs by floating calcareous algal mats and how that varied with distance from a canal gate. We found a threshold response in mat cover at a 7.76 km from the canal gate discharge into a marsh, with very low mat cover of sloughs close to the gate and very high mat cover farther than 7.76 km (Stevenson et al. 2002). A CART model explained much more variation than a linear regression model in the SSR between percent mat cover and distance to the canal gate, which provided credibility for classifying this response as a threshold and for using this benchmark as a restoration target. Based on models of water-column TP as a function of distance from the canal, TP was predicted to be about 10 μg TP $\cdot$ L^{-1} at 7.76 km from the canal gate.

A threshold response in periphyton species composition was also observed about 7.75 km from the canal inputs (McCormick et al. 1996, Pan et al. 2000, Payne et al. 2000). Similar responses of calcareous periphyton were observed along other phosphorus gradients in the Everglades by Gaiser et al. (2006). The 75th percentile of TP concentrations at reference sites in the Everglades was 10 μg $\cdot$ L^{-1}. This TP concentration was lower than concentrations showing effects in experiments. Due to dissimilarities in scale of experiments and the phosphorus gradient across marshes, differences in phosphorus forms added in experiments and in marshes exposed to canal water, and to long-term equilibrium in

phosphorus biogeochemistry and biological response in marshes, the benchmark for TP to sustain floating calcareous algal mats was based on SRRs derived from TP gradients across the marshes. Changes in macrophyte and invertebrate species composition were also observed along TP gradients, but their threshold responses were at higher phosphorus concentrations than loss of floating calcareous algae (King and Richardson 2003, McCormick et al. 2009).

Based on loss of floating calcareous algal mats, other changes in "natural populations of flora," and phosphorus concentrations at minimally disturbed sites in the Everglades, 10 μg TP · L^{-1} was selected by Florida's Department of Environmental Protection as the criterion to protect the Everglades. Although multiple lines of evidence were used to establish the TP criterion, the loss of the floating calcareous algal mat alone was a strong indicator of an "imbalance in the natural populations of aquatic flora and fauna." The 2003 Amendment to the Everglades Forever Act upheld the 10 μg · L^{-1} TP criterion and called for the earliest possible compliance. Today, challenges remain with achieving the 10 μg TP · L^{-1} criterion because innovative treatment technologies are needed to reduce phosphorus in canal waters to such low concentrations.

MANAGEMENT

Water resource protection and restoration are informed by algal bioassessments and accompanying ecological assessments that identify problems with algae that need to be solved, the stressors causing the problems, as well as the sources of stressors. Ecological assessments enable identification of current problems and vulnerabilities to future problems. Stressor criteria are important as targets for management of stressor levels. SRRs provide models predicting certainty of resource protection and responses to be expected during restoration (Stevenson 1998). Depending upon current condition and SRRs, restoration may result in immediate effects or require greater stressor reduction and longer times to produce positive results. Assessments of land use in watersheds provide a valuable first step for identifying sources of stressors. Identification and quantification of stressor sources provides options for stressor reduction to which specific costs can be assigned. Alternatively, as in the case of the Everglades, new technologies may need to be developed to achieve goals of the restoration.

Selecting the appropriate stressor management options depends on the costs and benefits of restoration as well as existing condition. The costs of management are associated with construction and operation of management options. The benefits of management for human well-being come from protecting and restoring EGS. Algae directly,

indirectly, positively, and negatively affect EGS. The values of aquatic EGS and their alteration by stressors affecting algae have been quantified and shown to be substantial (Dodds et al. 2009), even when corrected for distinctions between intermediate and final EGS. Final EGS have value because they contribute directly to human well-being. Therefore, intermediate EGS have indirect value because they support or regulate final EGS, so they could be assigned value based on their contribution to supporting final EGS but not added to the final EGS value to measure cumulative value of EGS.

EGS and their valuation will become more important in ecological assessments as pressures on environmental management increase with human population density, economic globalization, transboundary and global dispersal of pollutants, and resource limitation. EGS and their valuation are an international currency for environmental management. Variation in culture, economic conditions, and available resources affect regional valuation of EGS, thereby challenging arguments for international policy to protect EGS, such as biodiversity and the ecological integrity to protect biodiversity, clear lakes and rivers, and wetlands, which may not be a priority for the majority of people in many cultural and economic regimes.

The USEPA initiated pilot research projects on assessment and management of EGS. Managing algal conditions provides a good example of the challenges for resource managers because tradeoffs exist for managing aquatic EGS along nutrient gradients. Nutrients positively affect food webs and some provisioning ecosystem services and negatively affect water clarity, dissolved oxygen, water chemistry, and related drinking water, aesthetic, and cultural EGS (including biodiversity). In addition, EGS derived from economic activities that produce nutrient pollution in urban and agricultural ecosystems are important in the management model. Historically, research has focused on ecological challenges posed by managing systems for minimally disturbed conditions, which dominated the policy paradigm of federal agencies around the world. In contrast, resource managers face challenges for managing waters for the diversity of their "uses" without major compromises to economic activities. To optimize natural resource and economic performance across a region, as well as sustainable resource use, resource managers need more quantitative understanding of SRRs between EGS, metrics commonly used to assess ecosystems, and stressors. In addition, monetary and nonmonetary valuation of EGS and their support of human well-being will help optimize natural resource and economic performance to support human well-being. Policies that allow tiered uses of waters and managing different waters for different uses provide the regulatory mechanism for integrating ecosystem

services and ecological condition into a water policy framework.

CONCLUSIONS AND FUTURE DIRECTIONS

Assessment of the structure and function of algal communities and understanding of algal ecology has advanced greatly in the past two decades with wider application in state and federal programs. The USEPA collects phytoplankton and sediment diatoms in the National Lake Survey and benthic algae for the National Rivers and Stream Survey and National Wetland Condition Survey (USEPA 2009, 2013). Many states in the United States also use algae in assessments of streams and lakes (Bahls 1993, Danielson et al. 2011, 2012). Countries of the European Union and New Zealand use algae in assessment of lakes, rivers, and streams (Biggs et al. 1998, Rott et al. 2003, Kelly et al. 2009, Poikane et al. 2011, Birk et al. 2012). Research and application of algal bioassessment have also advanced in South Africa, China, India, Brazil, Argentina, Ecuador, and many other countries around the world. Why? Human activities alter ecological systems in ways that greatly affect algae and algae are excellent indicators of ecosystem change.

Water resources are among the most valuable and most widely threatened. Sound science is needed to wisely manage these resources with appropriate balance between over protection and under protection. The importance of tradeoffs among ecosystem services in watershed management is sufficiently great that research is needed for highly refined quantitative relationships that address ecological complexity, complexity of CHANS, and environmental policy. Proposals have been advanced for including degradation of ecosystem services as a key element in international environmental policy (National Academy of Science 2012). Given the great international variation in valuation of ecosystem services with culture, economic conditions, and geoclimatic setting, these refined understandings of complexity in CHANS will provide models that will resolve disputes of fairness based on local versus international valuation of ecosystem services (Stevenson 2011).

Large-scale assessment programs are providing data with sufficient detail, sample size, and scale to fuel an explosion in ecological knowledge. For example, the USEPA has ongoing National Aquatic Resource Surveys in which algae are characterized in 1,000s of lakes, streams, rivers, and wetlands. The National Aquatic Resource Surveys provide broad spatial coverage, but sampling is limited to one time. The United States Geological Survey characterized algae in hundreds of rivers and streams with one-time sampling and repeated sampling at a selected subset of sites, thereby enabling assessment of temporal as well as spatial patterns. The United States National Science Foundation will characterize algae in a relatively small number of strategically selected lakes, rivers, and streams in their National Ecological Observatory Network (NEON) program, which will emphasize temporal variability with repeated sampling at sites within and across multiple years. The data from these surveys are being made available to researchers with the explicit goal of enabling further analysis and interpretation of data and use of data in complementary research projects. In combination with experimental and modeling approaches, these survey data should greatly advance our understanding of algal ecology and advance new concepts in algal assessment and aquatic resource management.

We need to better understand the effect of human activities and resulting global change on the biodiversity of algae and significance of loss in biodiversity for EGS. We know that taxonomy based on morphology alone is not telling us everything we need to know about algal biodiversity (Manoylov 2014). In addition, our sample analysis protocols grossly underestimate species richness in assemblages. Loss of algal biodiversity could affect ecosystem function (Cardinale et al. 2006). Changes in dominant species in ecosystems certainly alter EGS. Evolution of algae may not be sufficiently rapid to adapt to global change and support important ecosystem functions (Thomas et al. 2012). If we are to protect resilience of algal function in aquatic ecosystems, can this be accomplished by protecting high levels of biodiversity in a subset of ecosystems? If so, which ecosystems and how many ecosystems must be protected to maintain resilience of algal function in aquatic ecosystems?

Algal biologists need to work more closely with economists and social scientists, as well as biogeochemists, hydrologists, engineers, and policy makers, to better understand how their research can be related to valuation of ecosystem services, developing management strategies, and informing environmental policy. What is the incremental improvement in algal condition, EGS, and EGS value with incremental reductions in stressors? Do thresholds exist in relationships among algal condition, EGS, and EGS valuation? What is the best way to deliver information to stakeholders and inform policy making? What new analytical methods are needed to address existing problems and future problems? We need cross-disciplinary collaborations and transdisciplinary integration and advancement of knowledge to inform policy making as effectively as possible.

As a wide variety of disciplines in ecology, engineering, economics, and social sciences have been applied to solve problems over the past 4 decades, they have expanded and converged on a set of common questions. How have humans altered ecosystems? Which alterations do we need to fix? How can we fix them? Great progress has been made in each of these disciplines. Great progress has been made with interdisciplinary collaborations. This progress

has been marked by the successive emergence of new names for disciplines associated with this progress. Systems science spun off from environmental science and engineering in 1970s, then evolved through stages of biocomplexity, conservation biology, restoration ecology, complex adaptive systems, couple human and natural systems, and finally sustainability science. As demand increases for environmental management, a new social contract for science has been written for this "Century for the Environment" (Lubchenco 1998). Algal ecology, taxonomy, systematics, physiology, biochemistry, and engineering will continue to be critical sciences for managing aquatic ecosystems, particularly within the broader transdisciplinary context of environmental and sustainability sciences.

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