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Taking the pulse of the ecosystem: progress in quantifying aquatic ecosystem health

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Abstract

Ecosystem health metrics quantify the cumulative effects of stressors on ecosystem structure and function, and inform management, restoration, and policy decisions. Freshwater ecosystems, in particular, face numerous stressors, and as a result, there is an increasing array of health metrics applied to their management. In this chapter, we review the current use of ecosystem health metrics, develop a preliminary framework for metric selection, and identify gaps in the current suite of metrics. The existing metrics typically characterize the biological, physical, or chemical attributes of ecosystems, whereas a few additional metrics integrate across these categories. Metrics vary in complexity, ranging from simple, visual assessments that can be completed by volunteers, to complex numerical models with extensive data and expert input requirements. Overall, ecosystem health metrics are well developed and useful with metrics available to fit both general and specialized management needs. However, common challenges include difficulty in establishing suitable reference conditions, a lack of uncertainty estimates, and a lack of inter-metric comparisons. Recent technological improvements, such as remote sensing, computational models, and new genetic sequencing techniques, are facilitating the development of novel and more holistic metrics, including early warning metrics, coupled complex systems models, and the inclusion of public input data.

Section 1. Introduction

Freshwater ecosystems face numerous stressors, including chemical pollutants, eutrophication, loss of biodiversity, and altered flow regimes (e.g., Carpenter et al. 1998; Milly et al. 2005; Diaz and Rosenberg 2008; Holtgrieve et al. 2011). Such stressors often negatively affect ecosystem structure and function, resulting in harmful algal blooms, proliferation of invasive species, declines in sensitive species, and reduced water quality. Whereas stressor effects on ecosystem structure and function are obvious in some cases, quantification of these effects is important to compare across systems and to identify sources of impairment. The quantification of ecosystem responses to stressors has been formalized through metrics that describe ecosystem health. These metrics have been used in a management context to set priorities for restoration and conservation, delineate impaired ecosystems, identify violations of clean water legislation, and to track success of water quality improvement efforts.

The concept of ecosystem health is popular because it is easy to understand metaphorically, yet from a practical standpoint, its definition remains somewhat vague. Consequently, a clear and defensible definition of ecosystem health is crucial for its effective application to the management of freshwater ecosystems. Whereas "ecosystem health" has been defined in numerous ways (e.g., Regier 1993; Karr and Chu 1999; Lackey 2001; Vugteveen et al. 2006), in this chapter we use the definition from Meyer (1997): A healthy ecosystem is "sustainable and resilient, maintaining its ecological structure and function over time while continuing to meet societal needs and expectations." We chose this definition because it includes ecosystem integrity, which is what most metrics attempt to quantify, and because it recognizes human needs and values in managing ecosystems.

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A broad range of metrics have been used to assess ecosystem health. They focus on characterizing the integrity of biological (e.g., Karr 1981), physical (e.g., Maddock 1999), and chemical (e.g., USEPA 2012) components of the ecosystem (Table 1). Some metrics integrate across these broad categories by combining physical, chemical, and biological measurements into a single index (e.g., Carlson 1977), whereas others use rates of ecosystem processes, such as gross primary production and respiration, as indicators of ecosystem health (e.g., Bunn and Davies 2000). Such a diversity of metrics and approaches may imply that we have a good understanding of the concept of ecosystem health and how to measure it. In fact, while numerous metrics have been developed, we lack a comprehensive, overarching framework for their application.

Choosing ecosystem health metrics poses many challenges. One must consider how well the metric characterizes ecosystem integrity, how it relates to the stressors involved, and whether it is appropriate for its intended usage—for example, is it appropriate to make management decisions based on a chosen metric? In particular, the application and interpretation of any metric must consider sampling frequency and location, natural variation in response metrics, the choice of appropriate reference conditions, and the temporal and spatial scale to which the results can be applied. Further complicating matters, the current set of ecosystem health metrics is not well-distributed across the components and attributes of aquatic ecosystems. For example, numerous metrics exist for fish and macroinvertebrate communities, whereas relatively few have been developed for ecosystem processes. As a result, it may be unclear which metric is most appropriate (e.g., how to choose the best fish metric), and in some cases, an appropriate metric may not be available (e.g., assessment of ecosystem function). Such challenges stem in large part from the lack of a universal framework to select the appropriate suite of ecosystem health metrics.

Our goal with this review is to move toward a more comprehensive understanding of ecosystem health metrics and the context in which they are applied. We begin by summarizing the state of knowledge on ecosystem metrics, and then develop a preliminary framework for metric selection by detailing appropriate metric usage and limitations across a range of freshwater systems. Finally, we identify gaps in the current suite of metrics and highlight aspects of freshwater ecosystems that are not currently characterized by any existing metrics. This discussion centers on metrics used in North America, but our general conclusions should be applicable across the globe. This synthesis provides a starting point for those interested in identifying appropriate metrics for their own systems, and also identifies avenues for new research where metrics are lacking.

Section 2. Overview and application of biological metrics

There is a broad suite of biological metrics used to assess ecosystem health in freshwater systems. They include numerous taxa, from bacteria to fish, and some metrics are applied simultaneously to multiple taxonomic groups (e.g., Pont et

Table 1. Categories of ecosystem health metrics.

Category	Type	Examples	Relevant ecosystems*	Ease of use	Limitations
Biological	MMIs	Index of biotic integrity (IBI)	S, L, W	Easy collection, processing can be lengthy	Defining reference conditions, linking stressors
	Single-parameter	Invertebrate functional groups	S, L, W	Easy and inexpensive	Defining reference conditions, linking stressors
	Organism health	Fish parasite or contaminant load	S, L	Specialized laboratory required	Linking results to ecosystem health
	Human health	*E. coli* population	S, L	Specialized laboratory required	Interpretation
Physical	Single-parameter	Secchi depth	S, L, W	Easy and inexpensive	Defining reference conditions, linking to stressors
	Visual surveys	Reach habitat assessment	S	Easy and inexpensive	Defining reference conditions
	Dynamical models	MesoHABSIM	S	Requires specialized expertise	Interpretation challenging
Chemical	Nutrients	N and P concentrations	S, L	Easy collection and processing	Defining impairment
	Emerging contaminants	Caffeine, pharmaceuticals	S, L	Specialized laboratory required	Effects on ecosystem health largely unknown
	pH, alkalinity	pH	S, L, W	Easy and inexpensive	Detects only specific stressors
Integrative	Multiple metrics	Trophic State Index (TSI)	L	Easy collection and processing	Defining reference conditions
	Ecosystem process	Metabolism	S, L	Easy collection, interpretation difficult	Defining reference conditions, defining impairment

*S = streams, L = lakes, W = wetlands

al. 2009), whereas others are taxon-specific (e.g., USEPA 2010, 2013). In addition, the structure of the biological metrics varies, and includes both multi-parameter (multivariate, multi-metric) and single parameter indexes. Multi-parameter metrics create an index score from multiple measurements, whereras single parameter metrics focus on a single indicator. Multivariate and multimetric indices primarily differ in whether site classifications are determined a posteriori (multivariate) with multivariate statistical methods (e.g., ordination, regression) or a priori (multi-metric) based on a suite of physicochemical and community metrics (Bowman and Somers 2005).

Many metrics are intended for a discrete area, such as a small stream reach or wetland, but others take a broader view and compare across sites (e.g., Soto-Galera et al. 1999). Whereas many biological metrics have been developed, the extent to which they have been applied across ecosystems remains uneven. For example, more metrics exist for streams than for lakes or wetlands. Additionally, there are other metrics that could be expanded to include more taxa (e.g., contaminant loading is currently focused on fish, but may be valid for other taxa).

Multi-metric indices (MMI) are the most extensively used biological metric to assess ecosystem health. These indexes, such as the Index of Biological Integrity (IBI, Karr 1981), incorporate numerous community-level parameters to generate a holistic picture of the communities and the quality of their environment. Examples of contributing metrics include species richness, presence of tolerant species, proportion of feeding guilds, and presence of rare taxa. Aquatic MMIs are most often used in streams and rivers (Ruaro and Gubiania 2013), but have also been adapted to lakes (Beck and Hatch 2009) and wetlands (Burton et al. 1999). MMIs typically include metrics that measure attributes relevant to a particular ecosystem type or to account for an intrinsic ecosystem characteristic (e.g., a species-poor system, Aparicio et al. 2011). In addition, the types of measurements included in MMIs depend on the taxonomic focus. For example, while initially developed for fish communities, IBIs have also been developed for macrophytes (Beck et al. 2010), plankton (Kane et al. 2009), invertebrates (Kerans and Karr 1994), and combinations of these groups (Pont et al. 2009). Although useful, MMIs are often developed for relatively localized areas based upon least disturbed or best available conditions (Bowman and Somers 2005), which complicates applying them in areas for which they were not designed. To overcome these limitations, recent efforts are focused on developing MMIs at regional, national, and continental scales (e.g., Stoddard et al. 2008).

Single-parameter biological metrics include community metrics, population parameters, multi-site comparisons, and harmful species. Community metrics characterize functional and taxonomic composition and are broadly used across taxa, including algae (benthic and phytoplankton), submerged aquatic vegetation (SAV), invertebrates, amphibians, and fish (Cummins and Klug 1979; Attrill and Depledge 1997; Poff et al. 2006; USEPA 2011a, Guzy et al. 2012). In addition, community metrics can focus on the proportion of the community that is native versus non-native (Davies et al. 2010). Population parameters include biomass, abundance, and density of an indicator species or taxon, and these metrics are used for algae, macrophytes, invertebrates, and fish (Hauer and Lamberti 2007; Davies et al. 2010; USEPA 2011a). An alternative approach is to infer ecosystem health from fish contaminant or parasite loads (Corsi et al. 2003; Palm and Ruckert 2009; Pietrock and Hursky 2011) or behavioral patterns (Gorman et al. 2012). Finally, some single-parameter metrics convey risk to humans via population counts of harmful species, including algae and bacteria (e.g., Wade et al. 2006). These metrics are not typically used explicitly as indicators of ecosystem health, although the proliferation of harmful species may indicate a larger ecosystem health problem, such as increased nutrient inputs.

Whereas most metrics are indicators of ecosystem health at a discrete location, some metrics encompass a larger spatial extent. These metrics address both the spatial pattern of ecosystem health and the magnitude of response to environmental conditions. Such analyses are conducive to large-bodied organisms and were developed for SAV and fish. SAV metrics include percent cover or habitat occupied, with the underlying asumption that macrophyte presence reflects good ecosystem health (USEPA 2011a). Fish metrics of spatial heterogeneity compare current versus historic distributions within and among watersheds, using changes in the proportion of native fishes and pollution-tolerant fishes to make inferences about spatial and temporal changes in ecosystem health among water bodies within a single watershed or among watersheds in a region (Soto-Galera et al. 1999). Contaminant load, which can be rapidly assessed by taking samples from fish at multiple locations, is also be used to characterize the spatial patterns of human impacts on aquatic ecosystems (e.g., Corsi et al. 2003). Metrics that incorporate multiple sites and spatial heterogeneity can allow for comparisons across a broader spatial scale, but they are not a common metric so far, perhaps because of their extensive data requirements or because there are fewer metrics that explicitly incorporate these comparisons.

Biological metrics are applied across stream, lake, and wetland habitats, but more metrics exist for stream habitats, both in terms of taxa use and type of metric. In general, invertebrate and fish metrics are considered integrators of whole-ecosystem condition due to their higher trophic status. In contrast, algae, macrophytes, and zooplankton often appear to be used for more narrowly-focused water quality and trophic state assessments. Amphibian-based biological metrics exist, but are less commonly implemented than metrics for other taxa (but see Welsh and Hodgson 2008 for an example; and Kroll et al. 2009 for a critique). Biological metrics, though diverse in form, strive to encompass multiple biological aspects of the ecosystem and provide numerous ways to assess ecosystem health.

Section 3. Advantages and disadvantages of biological metrics

Biological metrics are useful indicators for a number of reasons. They can be inexpensive to measure, completed rapidly with experts or volunteers, and can provide an informative assessment of the ecosystem. Furthermore, biological metrics can integrate through time and can detect environmental impacts that may not be readily detectable via physical or chemical assessments. Finally, numerous biological metrics exist, making it easier to find a metric suitable to specific monitoring or research goals across multiple spatial scales.

Biological metrics are also advantageous because they can be effective at monitoring the response of an ecosystem to short-term pulse disturbances. Pulse disturbances can be difficult to detect with direct measurements, because they require either 1) making real-time observations when the short-term event occurs, or 2) deploying expensive long-term monitoring equipment. However, community composition can shift in response to short-term disturbances, with declines in sensitive taxa and increases in tolerant taxa. In addition, many sensitive invertebrates, fish, and amphibians live for many years, and so their presence indicates consistently good water quality over their long lifespans, assuming they remain within a particular water body or a constrained stream reach.

A major strength of biological response variables are the number of biological metrics that can be measured rapidly and inexpensively. Rapid bioassessment protocols used by the EPA and state agencies allow for very quick and affordable assessments of stream ecosystems (USEPA 1990; Davis et al. 1996). With a low-cost sampling device such as a kick net and a basic understanding of aquatic invertebrate taxonomy, a team can rapidly grade an aquatic system. Fish IBI measurements initially require more expensive equipment (e.g., nets, backpack shockers, or shock boats), but can provide similarly efficient assessments. In addition to collecting field data, remote sensing data can provide macrophyte estimates to assess ecosystem health, which is also inexpensive. Remote sensing data require some ground-truthing and the appropriate skills and software to convert the data layers into usable coverage and density data.

Because biological metrics became more popular over the past 30 years, a number of advances have occurred, leading to more sophisticated metrics (Karr 1981; Karr et al. 1986; Ruaro and Guibiani 2013). For invertebrate-based metrics, many of these advances rely on identification of invertebrates to the lowest possible taxonomic resolution, which is often the species level. As a result, whereas improved taxonomic resolution can increase index reliability, it is often time-consuming and expensive and can require specialized personnel. Similarly, the accuracy of fish metrics improve with spatial extent surveyed, but increasing survey area may place practical limitations on equipment and labor. The advantages of biological metrics—speed and affordability—can disappear with improved accuracy. Thus, the need for improved metric accuracy must be weighed against budgetary constraints. The emerging field of metagenomics, identifying suites of species based on genetic techniques, may eventually resolve the trade-off between accuracy and cost.

An emerging technology that may improve biological metrics is the ability to detect environmental DNA (eDNA), which is DNA that was secreted or sloughed off by organisms living in a water body. Researchers can collect water samples, extract DNA, and compare it with molecular markers to determine if a target species is present (Jerde et al. 2011). This technique is more sensitive than survey methods, because it does not rely on physically encountering a species; DNA mixes well in the water column, whereas organisms tend to be found only in particular habitats (Pilliod et al. 2013). Furthermore, eDNA eliminates some of the problems with delineating sampling locations—while organisms may swim out of reach, eDNA remains in the water column for up to 2 weeks (Thomsen et al. 2012). So far, eDNA is successful in detecting certain rare (Goldberg et al. 2011; Pilliod et al. 2013), endangered (Thomsen et al. 2012), and non-native species (Jerde et al. 2011), but it has not yet been developed for IBIs or other multi-species metrics. Detection of eDNA is still a new technique, and many methodological details must be worked out before it is applied to multi-species metrics. For example, its use in IBIs will be limited by the fact that eDNA does not, as of yet, detect organismal abundance, just presence or absence. Nonetheless, eDNA techniques may improve the sensitivity of traditional survey methods, while decreasing labor and cost (Thomsen et al. 2012).

The application of a metric to the ecosystem of interest must also be carefully considered. In many cases, biological metrics are developed for particular regions, and there is a risk that a metric will perform poorly when it is used elsewhere. Furthermore, ecosystems that differ from the idealized healthy ecosystems on which models are based will perform poorly. For example, stream IBIs that include fishes often consider cold-water species, such as trout, indicative of a healthy ecosystem (Wang et al. 2003). Therefore, warm-water streams that lack native cold-water species will perform poorly with such a model. As with most metrics, care must be taken in the choice, application, and interpretation of biological metrics.

Another disadvantage of biological metrics is the assumption that biological integrity is closely linked with environmental quality, or that the biological community accurately reflects the current underlying physical and chemical conditions in a site (Karr 1981). For this assumption to be correct, the resident community must respond instantaneously to environmental changes, but often there is a time lag before perturbed communities stabilize. Consequently, it is critical to determine whether such time lags are large enough to affect the ability of biotic metrics to characterize ecosystem health. The longer the lag time in biotic response, the greater the likelihood that sampling will occur during a transitional point in the resident community; therefore, the greater the likelihood that the sampling does not accurately reflect current ecosystem health.

Meta-community ecology recently put an increased emphasis on the role of dispersal in community dynamics (Leibold et al. 2004). Dispersal can interfere with the interpretation of biological metrics in two ways. First, if dispersal limitation occurs and is unrelated to ecosystem health, we may underestimate the ecosystem health due to a time lag between physicochemical recovery of the system and the return of sensitive indicator species (Patrick and Swan 2011; Bogan and Boersma 2012). The alternative mechanism, mass effects, occurs when an unhealthy system receives inputs of sensitive species from a nearby healthy system, artificially improving its biological metric scores (Brown and Swan 2012). The mere presence of highly sensitive taxa does not necessarily mean that they are able to persist in that habitat and complete their life cycle. As a result, scores on biological metrics should be interpreted in light of physicochemical and mass effect conditions.

Biological metrics exist for many taxa and ecosystems, and there are numerous options for characterizing biological ecosystem health. However, the nature of biological communities can make these metrics difficult to interpret. In addition to the challenge of defining reference conditions, biological metrics must be interpreted in light of other confounding factors such as those that affect dispersal, time-lags in response to improvements in local conditions, and consideration of whether the metric is appropriate for the region and water body type. Overall, biological metrics provide a strong initial assessment of ecosystem health, but must be interpreted carefully and, ideally, within the context of an ecosystem's physical and chemical template.

Section 4. Physical metrics

Physical metrics include simple single-parameter measurements (e.g., water clarity, Wetzel 2001), but are often multi-parameter metrics that describe attributes of the water and geomorphic setting of aquatic ecosystems. The multi-parameter metrics typically combine several elements of the physical environment, such as slope, flow, water depth, and shoreline characteristics, to provide a single index. Techniques vary from rapid visual inspection of flow and geomorphic conditions (e.g., Pfankuch 1975; Rankin 1989) to modeling of numerous system parameters (e.g., Bovee et al. 1998; Parasiewicz and Dunbar 2001; and Milhous and Waddle 2012).

The spatial and temporal dynamics of the physical habitat are determined by the interaction of the structural features and hydrological regime of the aquatic ecosystem (Maddock 1999). Therefore, the physical conditions used to assess ecosystem health often focus on quantifying the characteristics of the flux and storage of water and sediment, as well as the geomorphic aspects of lotic and lentic environments. In many cases, the acceptable range of physical conditions is defined by the documented habitat requirements for a targeted individual species (e.g., seasonal fish habitat requirements, Gillette et al. 2006), and thus are often used in concert with biological metrics. In some cases, physical metrics are able to explain the absence of certain species even if other chemical and biological metrics indicate high ecosystem health.

Physical metrics are most commonly applied in streams, where physical processes can have a strong effect on habitat and ecosystem function, and where the water levels can change rapidly enough to have a strong effect on biological responses. They are also applied in lakes and wetlands, although fewer lentic metrics exist, and they are generally less complex.

In streams, some physical metrics are measured by field-based surveys that primarily draw upon visual assessments of ecosystem conditions, including the Qualitative Habitat Evaluation Index (QHEI), the Environmental Monitoring and Assessment Program (EMAP West), the Pfankuch channel stability, and the Vermont Agency of Natural Resources (VANR) Reach Habitat Assessment (RHA) (Pfankuch 1975; Rankin 1989; VANR 2008; Stoddard et al. 2005). The VANR RHA is a good example of a field-based survey, and it includes rapid assessments of woody debris cover, stream bed substrate cover, scour (pool) and depositional (riffle) geomorphic features, reach channel morphology, flow regime characteristics, surface flow connectivity, river banks characteristics, and riparian area conditions (VANR 2008). In general, field-based visual surveys provide a relatively fast way to evaluate and combine numerous ecosystem elements into a single index.

An alternative approach is dynamical and quantitative ecosystem health assessment methods, which are emerging from research on environmental flows (e.g., Poff et al. 1997; Dyson et al. 2003; Arthington et al. 2006) and ecogeomorphology (e.g., Wheaton et al. 2011). These methods couple empirical and theoretical approaches through the development of simulation models (e.g., MesoHABSIM, Parasiewicz 2008; and PHABSIM, Milhous and Waddle 2012). These models are typically parameterized and calibrated with detailed field and hydrological data, and then used to evaluate many aspects of aquatic ecosystem health, including habitat availability. In addition, many of these models explore how criteria for organisms in the stream vary with changes in the flow regime, geomorphic conditions, and habitat quality. The models are also capable of parameter sensitivity analysis, which can identify the key controls on habitat conditions across different hydrogeomorphic settings. In addition, dynamical models use potential parameter distributions, rather than single fixed values or categorical scores, and determine which combinations of parameters exceed ecosystem health thresholds (e.g., insufficient flows for a species of concern). These features of the model are then able to help refine efforts of field sampling and surveying. Overall, these models incorporate numerous physical attributes and predict responses to environmental change and management, while incorporating system variability.

In lakes, physical metrics include both single- and multi-parameter indexes. The most common lake physical measurement is likely the Secchi depth, an easy measurement of water clarity that can be completed by volunteers. In general, water quality increases with Secchi depth, because stressors such as eutrophication and erosion result in lower Secchi depths (Bruhn and Soranno 2005). More sophisticated optical analyses of lake waters, such as the suite of climate forcing optical indices (CFOI) proposed by Williamson et al. (2014), show promise as indicators of large scale climate change effects across a wide range of lake types. In contrast, most multi-parameter physical metrics involve visual assessments of shoreline habitat, in which lakes with natural vegetation, macrophytes, and natural banks score well, whereas lakes with manicured lawns, trash, and unnatural beaches score poorly (USEPA 2011a; McGoff et al. 2013). Often, the multi-parameter metrics are used in conjunction with biological data, where the physical metrics provide information on the physical stressors and habitat availability. Similarly, wetland physical metrics are often descriptive, and include soil profile and physical characteristics, water source and depth, and hydrological stressors (e.g., USEPA 2011b). They are typically used to determine the source of stress to biota or to provide information on the potential for wetland nutrient uptake or other ecosystem services, rather than functioning as independent indicators.

Section 5. Advantages and disadvantages of physical metrics

Survey-based physical assessments all share fundamental strengths and limitations. They can provide high-quality and easy-to-interpret data, which can guide management efforts, and ultimately reduce ecosystem stressors, especially those that affect organisms. Often, survey data are easy to obtain, and data collection can be completed by volunteers. Furthermore, they can indicate which activities and perturbations lead to long-term beneficial or harmful conditions in aquatic ecosystems (e.g., long-term RHA assessments that chronicle the effects of land-use change on sediment transport). The physical metrics in this type of assessment also provide important information about the potential for recovery of a degraded ecosystem upon alleviation of other biological and chemical stressors. Therefore, they provide information about the fundamental hydrogeomorphic template of an aquatic ecosystem and the ability of that template to support a healthy ecosystem.

Survey-based physical assessments also share common limitations in their ability to evaluate ecosystem health. In these surveys, many measurements are made of different categories of ecosystem health, where categories are determined a priori, and these measurements are converted to scores of overall health, based on a comparison with reference conditions. These scores may vary between practitioners and lead to subjective sources of uncertainty. In addition, inappropriate reference conditions can make conclusions questionable. Last, all of these survey methods require significant extant data on targeted species or ecosystem qualities to identify their fundamental ecosystem needs and tolerances. The distillation of many components to a single score is also a source of uncertainty. This uncertainty will continue to diminish as more studies and better understanding of a larger suite of species become available for aquatic ecosystems. Still, survey-based physical assessments can rapidly provide useful information on ecosystem health, as long as the results are interpreted with care.

Flow regime is a good indicator of habitat type and biodiversity potential in streams (Harper et al. 2000), so measuring, monitoring, and modeling flow and sediment conditions are key components of evaluating stream ecosystem health (Strange et al. 1999). The hydrogeomorphic modeling efforts require detailed field and hydrological data to parameterize and calibrate the models, but once the model is developed, many aspects of aquatic ecosystem health, including habitat availability, can be evaluated within the model system. This is particularly advantageous over the survey-based assessment methods, because they are capable of simulating different conditions in the ecosystem (e.g., flow and sediment regime changes) and determining the impact of various management scenarios on ecosystem features, such as fish habitat (e.g., Parasiewicz 2008).

Aquatic ecosystem habitat models were previously limited by computational demands and the ability to characterize and parameterize the model domain (Maddock 1999). However, new advances in computing and remote sensing techniques are removing these limitations. For example, the increasingly regular use of LIDAR (Laser Interferometry Detection and Ranging) and UAV (unmanned aerial vehicle) remote sensing technology in watershed surveys and river habitat mapping will increase time-efficiency and allow for the linking of geographic measurements across scales. Many small streams can have their morphology and flow conditions well-characterized with LIDAR at high spatial and temporal resolutions (e.g., Wheaton et al. 2013), which makes habitat model (e.g., MesoHABSIM) parameterization much less difficult and more precise than field ground-survey methods. LIDAR is a particularly exciting technology because it is a single tool that can map micro-habitat to watershed scales, which offers the opportunity to integrate physical ecosystem health metrics and processes across numerous scales. Whereas physical habitat models require substantial investment in time and resources, their predictive power and flexibility make them a powerful tool. Thus, they may be particularly useful in high-profile sites or those that have special management concerns.

Section 6. Chemical metrics of ecosystem health

Chemical metrics are based on the concentration of compounds in aquatic systems, including both naturally produced and synthesized compounds. These metrics can be divided into three categories: nutrients and organic compounds; human-synthesized compounds; and pH and alkalinity. The most common chemical metrics used to assess freshwater ecosystem health are dissolved inorganic nitrogen (DIN), including ammonium (NH_4^+) and nitrate (NO_3^-), total nitrogen (TN), phosphate (PO_4^{3-}), and total phosphorus (TP). Nitrogen and phosphorus are related to ecosystem health because they often limit biotic production in rivers, lakes, wetlands, and estuaries (Schindler 1977; Howarth and Marino 2006; Elser et al. 2007). Recently, dissolved organic matter (DOM) rose in prominence as an indicator of aquatic ecosystems health (e.g., Roulet and Moore 2006). Together, excess nutrients and organic matter enrichment are two of the leading causes of freshwater impairment in the United States (USEPA 2012). Although DIN and PO_4^{3-} are present due to natural processes (e.g., biogeochemical transformations, weathering), human activities have greatly accelerated their loading into freshwaters (Vitousek et al. 1997; Smith et al. 1999). Elevated nutrient concentrations from human activities may not only affect the freshwater ecosystems in human-dominated landscapes, but in remote ecosystems as well (Holtgrieve et al. 2011).

Regulatory agencies establish region-specific standards for inorganic nitrogen and phosphorus that vary in order to account for natural variation in climate and geology (e.g., USEPA 1998; USEPA 2000a). For example, the United States Environmental Protection Agency (USEPA) established criteria across 14 ecoregions for lakes and reservoirs of the US (8–38 μg L^{-1} of total P, 0.1–1.3 mg L^{-1} of total N) as well as for streams and rivers (10–76 μg L^{-1} of total P, 0.2–2.2 mg L^{-1} of total N) (USEPA 2000a). These criteria are often highest for the ecoregions that are most heavily impacted by humans (e.g., Corn Belt/Northern Great Plains of the US) and lowest in ecoregions with more forests and less agricultural land (e.g., Western Forested Mountains of the US) (e.g., USEPA 2000b).

Manufactured compounds in freshwaters, such as pharmaceuticals, pesticides, and nanoparticles, can cause adverse effects on ecosystem and human health (e.g., Schwarzenbach et al. 2006; Griffitt et al. 2008). Along with N and P, the USEPA lists pathogens, mercury, and heavy metals as the primary causes of freshwater impairment in the US (USEPA 2012). Freshwater ecosystems in human-dominated landscapes are often the most impacted by man-made compounds that contaminate ecosystems via sewage, industrial waste, and intensive agricultural practices (e.g., Buerge et al. 2003). However, man-made compounds have also been recorded in relatively pristine ecosystems (e.g., Mast et al. 2007), which suggests that monitoring freshwater ecosystems for such compounds should not be restricted to human-dominated landscapes. In addition, these compounds can be found in the water column and stored in sediments, so their persistence in the environment may be extensive. The ecological effects of man-made compounds are as diverse and varied as the compounds themselves, ranging from the relatively short-term, innocuous effects of elevated caffeine on stream biofilms (Lawrence et al. 2012) to the long-term, toxic effects of heavy metals on freshwater biota (Schubert et al. 2008). Risk assessments and ecotoxicology studies are becoming more common, but standards for many of these compounds are not yet established. Rather, the presence of one or several compounds can indicate that the health of the ecosystem is compromised (e.g., Nakada et al. 2008).

The pH and alkalinity of freshwaters is a third category of chemical metrics used to assess freshwater ecosystem health. Human impacts, such as mining and industrial emissions, can elevate the concentration of hydrogen ions (H^+) in freshwaters via both direct and indirect inputs (e.g., acid mine drainage and NO_X and SO_X emissions leading to acid rain, respectively) (Driscoll et al. 2001). Research has demonstrated that the biological community and overall ecosystem function are impaired in freshwaters with unnaturally low pH (e.g., <5), compromising the development and/or physiology of freshwater organisms and altering the ecosystem's buffering capacity (Schindler et al. 1985; McCormick et al. 1994; Likens et al. 1996; Niyogi et al. 2002; Kaushal et al. 2013). The use of pH as a metric of freshwater ecosystem health is often targeted in regions where the impact of humans will most likely cause acidification (USEPA 2005). Because of the strong relationship between freshwater acidification and human activities (e.g., mining, industry, atmospheric deposition of NO_X and SO_X), regulatory agencies of many countries have established guidelines for these activities in an effort to reduce the acidification of freshwaters (e.g., Title IV of the 1990 amendments to the US Clean Air Act).

Section 7. Advantages and disadvantages of existing chemical metrics

Nutrient concentrations in freshwaters are robust metrics for assessing freshwater ecosystem health. Whereas rapid bioassessments may indicate ecosystem impairment, measurements of inorganic N and P may indicate the potential mechanism of the observed impairment (USEPA 1991). Interpretation of N and P data are straightforward and the results can be compared easily across space (within a single water body and among water bodies regionally) and time. However, N and P concentrations fluctuate on various time scales due to storm events, biological uptake, watershed characteristics, or seasonality (Welter et al. 2005; Petrone et al. 2006). Therefore, sampling at regular intervals or under similar conditions (e.g., baseflow in streams, seasonal turnover in lakes) is most effective for relating these concentrations to ecosystem health. Laboratory facilities are necessary for accurate measurements, but water sampling kits are available for coarse measurements, allowing local municipalities or citizen scientists to assess N and P concentrations of their local freshwater ecosystems (Thornton and Leahy 2012).

The USEPA has established nutrient criteria for freshwaters within the conterminous US (USEPA 2000a), which is a helpful starting point. However, the approach used by the USEPA to assign nutrient criteria to each ecoregion is one of several, each of which may suggest different criteria within an ecoregion (Herlihy et al. 2013). Other techniques used to establish criteria and/or reference conditions for an ecoregion include paleolimnological reconstruction (e.g., Herlihy et al. 2013) or, in human-dominated landscapes where few reference ecosystems exist, a covariance approach among impaired ecosystems (Dodds and Oakes 2004) or an ecosystem classification approach (Soranno et al. 2010). Furthermore, the established nutrient concentrations provide a useful guideline, but ecosystems below these concentrations may still be impaired. For instance, TN concentrations above 1 mg L^{-1} suggest human impact on a stream or river, but these ecosystems would not be considered impaired given the criteria established for some ecoregions (e.g., Corn belt of the US; USEPA 2000b). Therefore, nutrient concentrations should only be used as one of several tools used to assess the health of an ecosystem.

The limitations of N and P as metrics of ecosystem health are primarily a function of variability in freshwater ecosystems. Whereas measuring concentrations of certain solutes is relatively straightforward and inexpensive, simply comparing the data to an established standard may not accurately assess ecosystem health. For instance, previous studies demonstrated that lakes are often P-limited and marine estuaries N-limited (e.g., Howarth and Marino 2006), but these generalizations are broad and do not apply to all lake or estuarine ecosystems (Elser et al. 2007). Furthermore, a given freshwater ecosystem may be more or less sensitive to small changes in the concentration of a specific solute due to the physical and/or biological characteristics of that particular system. Therefore, similar changes in a solute concentration may have a disproportionately large effect in some ecosystems, but not in others. Because broad nutrient criteria may not be applicable to all freshwater ecosystems, more nuanced measurements may provide the detailed information necessary for the appropriate management of specific ecosystems, such as the more integrative nutrient limitation and demand studies discussed below.

The standards established for man-made compounds are nearly as diverse as the compounds themselves, with the standards varying according to the toxicity of that compound on biota and its residence time in the environment (e.g., Schwab et al. 2005). The majority of standards that do exist are established for human, not ecosystem, health and expressed as tolerable daily intake (TDI; USEPA 1989). However, there are many groups of compounds present in freshwaters that likely pose a risk to human and ecosystem health (e.g., pesticides, such as diazinon and dieldrin), but have no TDI or similar standards established (Murray et al. 2010). Often, the presence of emergent pollutants alone would indicate that the health of the ecosystem is impaired (e.g., Buerge et al. 2003), but more research is needed to establish appropriate standards and regulation of these compounds. In general, the use of man-made compounds as assessment tools for freshwater health may not be practical due to the lack of standards and measurement techniques, the variety of compounds present, the extent to which these compounds may become buried in or remobilized from sediments, and the highly variable concentrations of these compounds in time and space (e.g., Kolpin et al. 2002). Furthermore, the analysis of water or sediment samples for many of these compounds is often costly (Schwarzenbach et al. 2006), which makes broad temporal or spatial sampling and monitoring impractical.

A further disadvantage for using man-made compounds as indicators of ecosystem health is that many unknowns exist in regards to the rapidly increasing number of these compounds and their interactive effects. For instance, the combination of certain pharmaceuticals in aquatic ecosystems can have synergistic effects on ecosystem health (e.g., Cleuvers 2003). Pharmaceuticals are often challenging to understand because of high variation in their chemical characteristics, which may vary for the same compound (i.e., polymorphism) or change following human metabolism (Cunningham 2008). Though many man-made compounds may adversely affect freshwater ecosystems, research on their effects on freshwaters may never catch up to novel compound development (Deblonde et al. 2011). A possible solution is to prioritize efforts toward certain classes or groups of these compounds, where the potential ecosystem effects may be greatest due to high toxicity or widespread prevalence (e.g., Sanderson et al. 2004; Murray et al. 2010). Ultimately, the development of environmentally benign compounds will be the most effective way to minimize the impacts of man-made compounds on ecosystem health (Schwarzenbach et al. 2006).

Measurements of pH are a direct and straightforward assessment of ecosystem health. The pH of freshwaters is easy and inexpensive to measure. Low pH is often the result of chronic impairment, although some ecosystems are naturally more acidic than others. Freshwaters, in some cases, are able to recover from unnaturally low pH (e.g., Stoddard et al. 1999), but the recovery is often over long time scales, such as multiple years or decades. Therefore, regular monitoring may be useful to compare across ecosystems, whereas long-term monitoring is necessary to assess whether a system is becoming impaired or in recovery.

Section 8. Integrative metrics

Integrative metrics reflect the activities of multiple ecosystem components operating in tandem, providing a more holistic representation of ecosystem health. Because different environmental stressors may affect ecosystem structure or function to varying degrees, an approach that focuses on specific aspects of structure alone may misrepresent system health. In contrast, integrative metrics combine elements of ecosystem structure and function. There are two approaches to characterize ecosystem health in an integrative way. The multi-metric approach combines various physical, chemical, and biological measurements into a single synthetic index. In contrast, a process-based approach uses measurements of ecosystem processes that themselves integrate across multiple ecosystem components. Both approaches, however, attempt to measure functional aspects of ecosystem health directly, either by combining structural metrics associated with underlying drivers of function, or by measuring the ecosystem function itself more directly.

Integrative multi-metrics have been in use since the 1970s. Perhaps the most widely used is the trophic state index (Carlson 1977), which combines water clarity, nutrient concentration, and chlorophyll-*a* concentration to create a single metric of ecosystem productivity and health. Multiple metrics are also combined to reflect specific bioregional monitoring goals, as with the management of the Columbia River (Thom and O'Rourke 2005) or Laurentian Great Lakes (Bertram et al. 1999; Neilson et al. 2003; Shear et al. 2003). The selection of metrics is often made on the basis of specific management objectives (Pantus and Dennison 2005) to target the effect of specific stressors (Moss et al. 2003), or for cross-system comparative purposes (Dobiesz et al. 2010). The multiple-metric approach is useful because it reflects multiple sources of stressors (Karr and Chu 1999), although simplification to a single number may mask important differences among sites (Norris and Hawkins 2000).

Alternatively, ecosystem processes can be used as integrative indices of ecosystem health. Ecosystem processes are often driven by a wide range of physical, chemical, and biological factors, making them natural integrators of system health. For instance, leaf decomposition rates and organic matter retention have been used as an index of overall ecosystem health (Wallace et al. 1996; Quinn et al. 2007), in which a higher rate of decomposition and greater organic matter retention is indicative of a healthier ecosystem. In addition, nitrification (Hill et al. 2000) and nutrient uptake rates (Sabater et al. 2000) can be used as indicators of ecosystem health, with a more retentive, tightly-coupled system considered healthier. Perhaps the most fundamental of ecosystem processes, the creation and consumption of organic matter by primary production and respiration, are increasingly being used to assess ecosystem activity, as well as ecosystem health. These rates are often quantified using free-water measurements of dissolved oxygen (Staehr et al. 2010).

Whereas oxygen concentration or biological oxygen demand are established indicators of ecosystem health, especially with regard to wastewater discharge, only recently have metabolism measurements become widespread enough to be used as an indicator of ecosystem health. Measurements of ecosystem metabolism have most commonly been used to identify incidents of eutrophication in lakes, streams, and coastal margins (Oviatt et al. 1986; Smith et al. 2005; Matthews and Effler 2006; Kemp et al. 2009; Gucker et al. 2009). Metabolic rates also have been used to provide a holistic measure of the effects of physical disturbances, such as changes in flow regimes or turbidity in streams (Wiley et al. 1990; Floder and Sommer 1999; Young and Huryn 1999) or flooding in lakes (Tsai et al. 2005; Sadro and Melack 2012). Ecosystem metabolism has also been used as an indicator of changes in catchment processes associated with agriculture and industrial use (Wiley et al. 1990; Wilcock et al 1998; Young and Huryn 1999; Sanders et al. 2007; Williamson et al. 2008) or to demonstrate the effect of toxins or pollutants (Giddings and Eddlemon 1978; Laursen et al. 2002; Wiegner et al. 2003). Despite such widespread use, few studies use metabolism explicitly as a broad indicator of ecosystem health. This is partly due to the challenge of interpreting metabolic rates as indicators of ecosystem health, although some early attempts suggest that it is possible (Young et al. 2008).

Section 9. Advantages and disadvantages of integrative metrics

There are many advantages to using integrative metrics in assessing ecosystem heath. By incorporating a broader diversity of individual elements, they condense a wide variety of environmental factors (Niemeijer 2002), attempt to account for the complexity of aquatic systems and accommodate linkages between other components of the landscape. Ecosystem process metrics, such as nutrient uptake rates or ecosystem metabolism, provide a direct measure of ecosystem function. Integrative metrics, by virtue of incorporating multiple elements, should be less sensitive to small-scale environmental variability that complicates the interpretation of physical, chemical, or biological metrics. Technological and methodological advances have made measurement of many of these processes relatively straightforward. For example, ecosystem metabolism can be easily measured through deployment of automated sensors (Levi et al. 2013; Solomon et al. 2013). This technology allows for the continuous collection of metabolism-based metric data, increasing the temporal resolution of the data, and eliminating the problem of missing important ecosystem perturbation events or sampling during nonrepresentative conditions. However, the ease of making measurements does not remove the complications of interpreting such data in the context of ecosystem health.

Despite these advantages, few integrative metrics have received widespread use. Of those we have described, only trophic state index (TSI) is used regularly to monitor ecosystem health. It is an important component in water quality monitoring among many organizations, from the USEPA to individual lake or watershed associations (Carlson and Simpson 1996; USEPA 2000a). In a comparative analysis involving more than 30 lakes, Jorgensen et al. (2005) demonstrated a strong linear relationship between TSI and two more complex models of ecosystem health, suggesting that it does a good job of characterizing ecosystem health, despite its simplicity. Although not as widely used, some ecosystem process metrics are promising, including ecosystem metabolism in rivers (Young et al. 2008). The remaining metrics, which have primarily been used only in academic studies, have failed to gain traction largely because of the difficulty in translating measurements of ecosystem process to an index that can be interpreted across aquatic ecosystems and management schemes.

In addition, there are a number of complexities associated with the use of ecosystem process metrics. Two immediate challenges are 1) identifying which physical, chemical, and biological elements to include as proxies for ecosystem health (Costanza et al. 1992; Patil et al. 2001; Schaeffer et al. 1988), and 2) determining how to interpret them in a unified way. As with other metrics, there remains the issue of interpreting the condition of a specific site in the context of environmental variability, as well as the selection of reference sites (Dobiesz et al. 2010). Although ecosystem process-based metrics inherently provide an integrative assessment of ecosystem health, the myriad of factors that affect such processes, some of which may operate in opposition, make interpreting them in the context of ecosystem health difficult without a complete understanding of the system (Young et al. 2008; Reuther 1992; Wiegner et al. 2003). For example, low rates of gross primary production could indicate impairment of biota or reflect a system with naturally low rates due to climate and geology (Young et al. 2008). Likewise, high daily variability in these metrics makes long-term data sets and seasonal averaging important when interpreting changes in system dynamics (Dobiesz et al. 2010; Staehr et al. 2010; Coloso et al. 2011). Despite the appeal of integrating across biological, chemical, and physical ecosystem components, process metrics require a greater expertise and understanding of an ecosystem, perhaps limiting their application by small or volunteer-based monitoring programs.

Section 10. Advantages and challenges across ecosystem health metrics

The decision to incorporate metrics into management is influenced by the effort and expertise required to collect and interpret metric data. Numerous metrics can be completed rapidly and inexpensively by volunteers and citizen scientists, which can increase the number of sites monitored and the frequency of assessment, compared with metrics that require expensive equipment and specialized personnel. In general, though, rapid assessments provide less information than the more sophisticated metrics. As a result, the goals of a monitoring effort must be carefully considered before choosing metrics. For example, simple visual assessment may be a useful first step that identifies ecosystems for further study. Similarly, sophisticated metrics might be reserved for sites with impending management decisions, those of high economic or ecological importance, or sites that are particularly high-profile, whereas the simpler metrics may be used more broadly to quickly characterize an array of sites.

Many of the well-established metrics are based on a comparison with reference conditions; thus, the choice of a reference system is crucial to the validity of the results. They rely on defining an undisturbed reference (or a more realistic "Least-Disturbed Condition," Stoddard et al. 2005) against which to evaluate an ecosystem of interest. Identifying and quantifying any reference system, let alone one relevant to the ecosystem of interest, is a source of uncertainty because of natural variation in ecosystem characteristics, such as physical stability, biodiversity, chemical concentrations, and rates of ecosystem processes. In addition, some systems may have no known or measurable reference conditions. This limitation has been addressed by metrics that combine theoretical and empirical approaches (e.g., environmental flows, Poff et al. 1997), by metrics that are based on human health outcomes (e.g., populations of harmful bacteria, USEPA 2010), and by ecosystem classification modeling (e.g., grouping water bodies by their relationship between land use and nutrient concentration, Soranno et al. 2010). The continued development of these approaches will be an important contribution to the quantification of ecosystem health.

Finally, caution must be applied in the interpretation of any individual metric. By necessity, each metric addresses a subset of possible ecosystem health indicators, each subject to its own uncertainties, biases, and assumptions. Ideally, before making management decisions, managers will apply several metrics, each addressing different aspects of ecosystem health. Such an approach will provide a more complete picture of ecosystem health, analogous to doctors using multiple tests to assess a patient's health (e.g., blood pressure alone is insufficient for determining health). In addition, different metrics may provide complementary information; for example, a biological assessment may reveal a preponderance of pollution-tolerant taxa, whereas chemical or physical assessments may reveal the cause of the biological impairment. As a result, it may be useful to consider an individual metric as nested within a larger health assessment. This approach has gained traction recently. For example, the USEPA's National Wetlands Condition Assessment (USEPA 2011b) and the National Lakes Assessment (USEPA 2011a) both measure numerous metrics of biological, physical, and chemical health. In doing so, they are able to identify sites with impaired biotic communities, as well as the causes of that impairment.

Section 11. Gaps in existing metrics

There are currently numerous metrics available to assess nearly all aspects of ecosystem health in aquatic systems. Some metrics have a more formally developed framework of assessment than others (e.g., biological versus integrative metrics), and for a number of specific metrics we have identified places where additional refinement of application and interpretation would be useful and places where metrics are lacking, including those that address the effects of emerging contaminants and pharmaceuticals. In addition to these specific gaps, limitations remain in the general use of ecosystem health metrics, including a lack of uncertainty estimates, lack of inter-metric comparisons, and lack of coordination of sampling efforts.

A lack of uncertainty estimates for most metrics further complicates their interpretation; the assignment of a single score for a given metric may be too parsimonious. Despite a lack of replication, there are a number of alternative approaches that may be used to estimate uncertainty.

Estimates could be calculated from the number of sampling dates. For example, if a chemical concentration is measured only once, the uncertainty associated with that value would be high. Similarly, it could be calculated by comparing individual components of an index, where individual components with high uncertainties would result in an overall higher uncertainty. Such an approach would report a range or distribution, rather than a single value. These uncertainty estimates themselves might have important management implications, where ecosystems with large uncertainties would possibly receive further study before management decisions were made. Interpreting uncertainty estimates along with metrics of ecosystem health may help managers predict the likelihood of improvement from intervention. However, communicating uncertainty to the public remains complex. Whereas it is easy to interpret a single number, the public is generally less familiar with parameter distributions. Thus, if policy decisions are based on such an approach, the results must be presented in a way that makes sense to decision-makers and the public.

Whereas different metrics can provide similar information, there may be large differences in the effort required. For example, Wallace et al. (1996) demonstrated that leaf decomposition rates provided similar results to the percent Ephemeroptera, Plecoptera, and Trichoptera (%EPT), but % EPT was less labor- and time-intensive. These comparative studies can help ascertain which metrics overlap and which are most informative. Ultimately, a collection of these studies can be used to determine the best way to allocate monitoring resources and to determine where a particular metric is most appropriate. Thus far, few such studies exist, but these comparisons may be a fruitful area of future research. One particularly interesting application would be to compare integrative metrics, such as ecosystem metabolism or nutrient uptake, with easier-to-measure metrics such as nutrient concentration.

Finally, the vast number of metrics used across a myriad of monitoring programs constrains large-scale analyses and intercomparisons of metrics. To some degree, this is inevitable, as some metrics may be more appropriate for particular locations or ecosystem types. However, a standardized set of protocols and a depot for data-sharing may result in analyses that reveal broad spatial patterns. For example, the Global Lakes Ecological Observatory Network (GLEON) was able to identify the drivers of ecosystem respiration in lakes as a result of data collected in a consistent, standardized manner through a coordinated sensor network (Solomon et al. 2013).

Section 12. Priorities for future research

In addition to addressing the gaps in current metrics, future research can use technological, computational, and interdisciplinary tools to move the field forward. Ideally, these efforts will move beyond existing frameworks to create new paradigms for ecosystem health assessments, including the development of early warning metrics, the explicit inclusion of public input, and the use of complex systems models. Improvements in these key areas may help improve ecosystem health assessments and ultimately improve our ability to manage aquatic ecosystems.

Assessments of ecosystem health are often completed after an environmental impact has occurred, but metrics that can provide early warning of declines—analogous to preventative medicine—are likely more effective at preventing damage to ecosystems (Boulton et al. 1999). Early warning metrics may be aided by the increasing availability of automated sensors, and research priorities in this area should focus on identifying the early warning signals. These warning signals may be similar to those from the current suite of metrics, such as critical levels of a contaminant or a low-oxygen threshold, but the high frequency measurements allow the development of novel metrics. For example, theoretical and experimental work with long-term data sets suggest that the variance associated with a specific metric, the return time of a metric to baseline levels after a perturbation, or changes in the autocorrelation of a metric's time series may serve as early warnings of regime shifts (Scheffer and Carpenter 2003; Carpenter et al. 2011; Batt et al. 2013). These metrics use more complicated statistics than most existing metrics, and thus they may require further refinement before they are used broadly; water resource managers may not have access to the necessary statistical tools and policy-makers may not understand the terminology. These problems are not intractable, however, and can likely be overcome with careful consideration of these difficulties.

Automated sensors can also capture pulse events, such as dips in dissolved oxygen, rapid changes in pH, or sudden increases in sediment or nutrient concentrations. An increase in frequency or duration of acute events may indicate problems before they become evident in biological surveys. This approach will require research into baseline conditions, to establish a healthy range and timing of pulse events. For example, some sites may regularly experience periods of low dissolved oxygen, during summer low-flow conditions, to which the biota are well-adapted. Sensors do not yet exist for all potential stressors, which limits the types of pulse events that can be detected. Nonetheless, the current sensors provide a good starting point for the development of metrics, and initial research efforts will help determine the utility of applying this approach more broadly.

Early warning metrics will only be useful if they invoke meaningful actions. In addition to knowing what an early warning signal looks like, practitioners must have sufficient system-specific knowledge to understand the causes of impairment and know how to mitigate them. For all metrics, the mitigation process may be smoother if citizen and stakeholder input are explicitly included in a monitoring program. After all, identifying a "healthy" or "unhealthy" system is only the first step; mitigation requires the cooperation of numerous stakeholders.

Inclusion of public input may improve the iterative process between monitoring and management, and deserves further research. So far, public input is included in integrative models, which include stakeholder preferences, the policy environment, and scientific knowledge (Croke et al. 2007). Similarly, scenario-based models predict ecosystem responses to alternative management scenarios (e.g., Xu et al. 2013; Einheuser et al. 2012). In these models, stakeholders can see how different public policy options will influence water quality. Public input could also be included in simpler ways, such as through a survey that establishes the expected usage of the system or acceptable changes in land use or policy. Research in this area may benefit from collaborations with sociologists and economists, who can identify the relevant metrics for citizen input.

Physical, chemical, and biological ecosystem health metrics are often presented as discrete ways of measuring and understanding ecosystem health—indeed, we make that distinction in this chapter—but often, it is useful to consider multiple types of metrics. Integrative metrics, as discussed above, are one way to incorporate multiple ecosystem elements, but another priority for future research is the coupling of complex systems models. There are currently good models for each aspect of ecosystem health, and combining them into a larger, mechanistic model may improve our understanding of the causes of impairment. These models are not likely to be used as metrics themselves but instead will be applied to impaired water bodies, with the intention of pinpointing the sources of impairment, and ultimately improving management. Development of these models will likely require the collaboration of community ecologists, ecosystem ecologists, geochemists, and computer scientists.

References

Aparicio, E., G. Carmona-Catot, P. B. Moyle, and E. Garcia-Berthou. 2011. Development and evaluation of a fish-based index to assess biological integrity of Mediterranean streams. Aquat. Conserv. Mar. Freshw. Ecosys. 21:324-337 [doi:10.1002/aqc.1197].

Arthington, A. H., S. E. Bunn, N. L. Poff, and R. J. Naiman. 2006. The challenge of providing environmental flow rules to sustain river ecosystems. Ecol. Appl. 16:1311-1318 [doi:10.1890/1051-0761(2006)016[1311:TCOPEF]2.0.CO;2].

Attrill, M. J., and M. H. Depledge. 1997. Community and population indicators of ecosystem health: Targeting links between levels of biological organization. Aquat. Toxicol. 38:183-197 [doi:10.1016/S0166-445X(96)00839-9].

Batt, R. D., S. R. Carpenter, J. J. Cole, M. L. Pace, and R. A. Johnson. 2013. Changes in ecosystem resilience detected in automated measures of ecosystem metabolism during a whole-lake manipulation. Proc. Nat. Acad. Sci. 110:17398-17403 [doi:10.1073/pnas.1316721110].

Boulton, A.J. 1999. An overview of river health assessment: philosophies, practice, problems and prognosis. 41: 469-479 [doi:10.1046/j.1365-2427.1999.00443.x].

Bowman, M. F., and K. M. Somers. 2005. Considerations when using the Reference Condition Approach for bioassessment of freshwater ecosystems. Water Qual. Res. J. Can. 40:347-360

Beck, M. W., and L. K. Hatch. 2009. A review of research on the development of lake indices of biotic integrity. Environ. Rev. 17:21-44 [doi:10.1139/A09-001].

———, ———, B. Vondracek, and R. D. Valley. 2010. Development of a macrophyte-based index of biotic integrity for Minnesota lakes. Ecol. Indic. 10:968-979 [doi:10.1016/j.ecolind.2010.02.006].

Bertram, P., H. Shear, N. Stadler-Salt, and P. Horvatin. 1999. Environmental and socioeconomic indicators of Great Lakes basin health, pp. 703-720. *In* D. Rapport, W. Lasley, D. Rolston, N. Nielsen, C. Qualset, and A. Damania (Eds.), Managing for healthy ecosystems. Lewis Publishers.

Bogan, M. T., and K. S. Boersma. 2012. Aerial dispersal of aquatic invertebrates along and away from arid-land streams. Freshw. Sci. 31:1131-1144 [doi:10.1899/12-066.1].

Bovee, K. D., B. L. Lamb, J. M. Bartholow, C. B. Stalnaker, J. Taylor, and J. Henriksen. 1998. Stream habitat analysis using the instream flow incremental methodology. U.S. Geological Survey Information and Technology Report 1998-0004. USGS.

Brown, B. L., and C. M. Swan. 2010. Dendritic network structure constrains metacommunity properties in riverine ecosystems. J. Animal Ecol. 79:571-580 [doi:10.1111/j.1365-2656.2010.01668.x].

Bruhn, L. C., and P. A. Soranno. 2005. Long-term (1974-2001) volunteer monitoring of water clarity trends in Michigan lakes and their relation to ecoregion and land use/cover. Lake Reservoir Manage. 21:10-23.

Buerge, I. J., T. Poiger, M. D. Muller, and H. R. Buser. 2003. Caffeine, an anthropogenic marker for wastewater contamination of surface waters. Environ. Sci. Technol. 37:691-700 [doi:10.1021/es020125z].

Bunn, S. E., and P. M. Davies. 2000. Biological processes in running waters and their implications for the assessment of ecological integrity. Hydrobiologia 422:61-70 [doi:10.1023/A:1017075528625].

Burton, T. M., D. G. Uzarski, J. P. Gathman, J. A. Genet, B. E. Keas, and C. A. Stricker. 1999. Development of a preliminary invertebrate index of biotic integrity for Lake Huron coastal wetlands. Wetlands 19:869-882 [doi:10.1007/BF03161789].

Carlson, R. E. 1977. A trophic state index for lakes. Limnol. Oceanogr. 22:361-369 [doi:10.4319/lo.1977.22.2.0361].

———, and J. Simpson. 1996. A coordinator's guide to volunteer lake monitoring methods. North American Lake Management Society.

Carpenter, S. R., N. F. Caraco, D. L. Correll, R. W. Howarth, A. N. Sharpley, and V. H. Smith. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecol. Appl. 8:559-568 [doi:10.1890/1051-0761(1998)008[0559:NPOSWW]2.0.CO;2].

———, and others. 2011. Early warnings of regime shifts: A whole-ecosystem experiment. Science 332:1079-1082 [doi:10.1126/science.1203672].

Cleuvers, M. 2003. Aquatic ecotoxicity of pharmaceuticals including the assessment of combination effects. Toxicol. Lett. 142:185-194 [doi:10.1016/S0378-4274(03)00068-7].

Coloso, J. J, J. Cole, and M. Pace. 2011. Short-term variation in thermal stratification complicates estimation of lake metabolism. Aquat. Sci. 73:305-315 [doi:10.1007/s00027-010-0177-0].

Corsi, I., M. Mariottini, C. Sensini, L. Lancini, and S. Focardi. 2003. Fish as bioindicators of brackish ecosystem health: integrating biomarker responses and target pollutant concentrations. Oceanol. Acta 26:129-138 [doi:10.1016/S0399-1784(02)01237-9].

Costanza, R., B. G. Norton, and B. D. Haskell (eds.). 1992. Ecosystem health: new goals for environmental management. Island Press.

Croke, B. F. W., J. L. Ticehurst, R. A. Letcher, J. P. Norton, L. T. H. Newham, and A. D. Jakeman. 2007. Integrated assessment of water resources: Australian experiences. Water Res. Manage. 21:351-373 [doi:10.1007/s11269-006-9057-8].

Cummins, K. W., and M. J. Klug. 1979. Feeding ecology of stream invertebrates. Ann. Rev. Ecol. System. 10:147-172 [doi:10.1146/annurev.es.10.110179.001051].

Cunningham, V. I. 2008. Special characteristics of pharmaceuticals related to environmental fate. pp 23-33. *In* K. Kummerer (ed.), Pharmaceuticals in the environment. Springer [doi:10.1007/978-3-540-74664-5_2].

Davies, P. E., J. H. Harris, T. J. Hillman, and K. F. Walker. 2010. The sustainable rivers audit: assessing river ecosystem health in the Murray-Darling Basin, Australia. Mar. Freshw. Res. 61:764-777 [doi:10.1071/MF09043].

Davis, W. S., B. D. Snyder, J. B. Stribling, and C. Stoughton. 1996. Summary of state biological assessment programs for streams and wadeable rivers. EPA 230-R-96-007. U.S. Environmental Protection Agency: Office of Policy, Planning, and Evaluation.

Deblonde, T., C. Cossu-Leguille, and P. Hartemann. 2011. Emerging pollutants in wastewater: A review of the literature. Int. J. Hyg. Environ. Health 214:442-448 [doi:10.1016/j.ijheh.2011.08.002].

Diaz, R. J., and R. Rosenberg. 2008. Spreading dead zones and consequences for marine ecosystems. Science 321:926-929 [doi:10.1126/science.1156401].

Dobiesz, N. E., and others. 2010. Metrics of ecosystem status for large aquatic systems—A global comparison. J. Great Lakes Res. 36:123-138 [doi:10.1016/j.jglr.2009.11.003].

Dodds, W. K., and R. M. Oakes. 2004. A technique for establishing reference nutrient concentrations across watersheds affected by humans. Limnol. Oceanogr. Methods 2:333-341 [doi:10.4319/lom.2004.2.333].

Driscoll, C. T., G. B. Lawrence, A. J. Bulger, T. J. Butler, C. S. Cronan, C. Eagar, and K. C. Weathers. 2001. Acidic deposition in the northeastern United States: Sources and inputs, ecosystem effects, and management strategies. Bioscience 51:180-198 [doi:10.1641/0006-3568(2001)051[0180:ADITNU]2.0.CO;2].

Dyson, M., G. J. J. Bergkamp, and J. Scanlon (eds.). 2003. Flow: The essentials of environmental flows. International Union for Conservation of Nature and Natural Resources (IUCN) [doi:10.2305/IUCN.CH.2003.WANI.2.en].

Einheuser, M. D., A. P. Nejadhashemi, S. P. Sowa, L. Wang, Y. A. Hamaamin, and S. A. Woznicki. 2012. Modeling the effects of conservation practices on stream health. Sci. Total Environ. 435-436:380-391 [doi:10.1016/j.scitotenv.2012.07.033].

Elser, J. J., M. E. S. Bracken, E. E. Cleland, D. S. Gruner, S. W. Harpole, H. Hillebrand, and J. E. Smith. 2007. Global analysis of nitrogen and phosphorus limitation of primary producers in freshwater, marine and terrestrial ecosystems. Ecol. Lett. 10:1135-1142 [doi:10.1111/j.1461-0248.2007.01113.x].

Floder, S., and U. Sommer. 1999. Diversity in planktonic communities: an experimental test of the intermediate disturbance hypothesis. Limnol. Oceanogr. 44:1114-1119 [doi:10.4319/lo.1999.44.4.1114].

Giddings, J. M., and G. K. Eddlemon. 1978. Photosynthesis/respiration ratios in aquatic microcosms under arsenic stress. Water Air Soil Pollut. 9:207-212 [doi:10.1007/BF00280707].

Gillette, D. P., J. S. Tiemann, D. R. Edds, and M. L. Wildhaber, 2006. Habitat use by a midwestern USA riverine fish assemblage: Effects of season, water temperature and river discharge. J. Fish Biol. 68:1494-1512 [doi:10.1111/j.0022-1112.2006.001037.x].

Goldberg, C. S., D. S. Pilliod, R. S. Arkle, and L. P. Waits. 2011. Molecular detection of vertebrates in stream water: A demonstration using Rocky Mountain Tailed Frogs and Idaho Giant Salamanders. PLOS One 6:e22746 [doi:10.1371/journal.pone.0022746].

Gorman, O. T., D. L. Yule, and J. D. Stockwell. 2012. Habitat use by fishes of Lake Superior. II. Consequences of diel habitat use for habitat linkages and habitat coupling in nearshore and offshore waters. Aquat. Ecosys. Health Manage. 15:355-368.

Griffitt, R. J., J. Luo, J. Gao, J. C. Bonzongo, and D. S. Barber. 2008. Effect of particle composition and species on toxicity of metallic nanoparticles in aquatic organisms. Environ. Toxicol. Chem. 27:1972-1978 [doi:10.1897/08-002.1].

Gucker, B., I. G. Boechat, and A. Giani. 2009. Impacts of agricultural land use on ecosystem structure and whole-stream metabolism of tropical Cerrado streams. Freshw. Biolo. 54:2069-2085 [doi:10.1111/j.1365-2427.2008.02069.x].

Guzy, J. C., E. D. McCoy, A. C. Deyle, S. M. Gonzalez, N. Halstead, and H. R. Mushinsky. 2012. Urbanization interferes with the use of amphibians as indicators of ecological integrity of wetlands. J. Appl. Ecol. 49:941-952 [doi:10.1111/j.1365-2664.2012.02172.x].

Harper, D. M., J. L. Kemp, B. Vogel, and M. D. Newson. 2000. Towards the assessment of 'ecological integrity' in running waters of the United Kingdom. Hydrobiologia 422/423:133-142 [doi:10.1023/A:1017072906760].

Hauer, F. R. and G. A. Lamberti (eds.). 2007. Methods in stream ecology. Elsevier.

Herlihy, A. T., N. C. Kamman, J. C. Sifneos, D. Charles, M. D. Enache, and R. J. Stevenson. 2013. Using multiple approaches to develop nutrient criteria for lakes in the conterminous USA. Freshw. Sci. 32(2):67-384 [doi:10.1899/11-097.1].

Hill, B. H., R. K. Hall, P. Husby, A. T. Herlihy, and M. Dunne. 2000. Interregional comparisons of sediment microbial respiration in streams. Freshw. Biol. 44:213–222 [doi:10.1046/j.1365-2427.2000.00555.x].

Holtgrieve, G. W., and others. 2011. A coherent signature of anthropogenic nitrogen deposition to remote watersheds of the Northern hemisphere. Science 334:1545-1548 [doi:10.1126/science.1212267].

Howarth, R. W., and R. Marino. 2006. Nitrogen as the limiting nutrient for eutrophication in coastal marine ecosystems: Evolving views over three decades. Limnol. Oceanogr. 51:364-376 [doi:10.4319/lo.2006.51.1_part_2.0364].

Jerde, C. L., A. R. Mahon, W. L. Chadderton, and D. M. Lodge. 2011."Sight-unseen" detection of rare aquatic species using environmental DNA. Conserv. Lett. 4:150-157 [doi:10.1111/j.1755-263X.2010.00158.x].

Jorgensen, S. E., R. Costanza, and F. Xu. 2005. Handbook of ecological indicators for assessment of ecosystem health. Taylor and Francis, CRC Press [doi:10.1201/9780203490181].

Kane, D. D., S. I. Gordon, M. Munawar, M. N. Charlton, and D. A. Culver. 2009. The Planktonic Index of Biotic Integrity (P-IBI): An approach for assessing lake ecosystem health. Ecol. Indic. 9:1234-1247 [doi:10.1016/j.ecolind.2009.03.014].

Karr, J. R. 1981. Assessment of biotic integrity using fish communities. Fisheries 6:21-27 [doi:10.1577/1548-8446(1981)006<0021:AOBIUF>2.0.CO;2].

———, K. D. Fausch, P. L. Angermeier, P. R. Yant and I. J. Schlosser. 1986. Assessing biological integrity in running waters. A method and its rationale. Illinois Natural History Survey. Special Publication No. 5.

———, and E. W. Chu. 1999. Restoring life in running waters: Better biological monitoring. Island Press [doi:10.1023/A:1005167831280].

Kaushal, S. S., G. E. Likens, R. Utz, M. L. Pace, M. Grese, and M. Yepsen. 2013. Increased river alkalinization in the Eastern U.S. Environ. Sci. Technol [doi:10.1021/es401046s].

Kemp, W. M., J. M. Testa, D. J. Conley, D. Gilbert, and J. D. Hagy. 2009. Temporal responses of coastal hypoxia to nutrient loading and physical controls. Biogeosciences 6:2985-3008 [doi:10.5194/bg-6-2985-2009].

Kerans, B. L., and J. R. Karr. 1994. A benthic index of biotic integrity (B-IBI) for rivers of the Tennessee Valley. Ecol. Appl. 4:768-785 [doi:10.2307/1942007].

Kolpin, D. W., E. T. Furlong, M. T. Meyer, E. M. Thurman, S. D. Zaugg, L. B. Barber, and H. T. Buxton. 2002. Pharmaceuticals, hormones, and other organic wastewater contaminants in US streams, 1999-2000: A national reconnaissance. Environ. Sci. Technol. 36:1202-1211 [doi:10.1021/es011055j].

Kroll, A. J., M. P. Hayes, and J. G. MacCracken. 2009. Concerns regarding the use of amphibians as metrics of critical biological thresholds: a comment on Welsh & Hodgson (2008). Freshw. Biol. 54:2364-2373 [doi:10.1111/j.1365-2427.2009.02245.x].

Lackey, R. T. 2001. Values, policy, and ecosystem health. Bioscience 51:437-443 [doi:10.1641/0006-3568(2001)051[0437:VPAEH]2.0.CO;2].

Laursen, A. E., S. P. Seitzinger, R. Dekorsey, J. G. Sanders, D. L. Breitburg, and R. W. Osman. 2002. Multiple stressors in an estuarine system: effects of nutrients, trace elements, and trophic complexity on benthic photosynthesis and respiration. Estuaries 25:57-69 [doi:10.1007/BF02696049].

Lawrence, J. R., B. Zhu, G. D. W. Swerhone, J. Roy, V. Tumber, M. J. Waiser, and D. R. Korber. 2012. Molecular and microscopic assessment of the effects of caffeine, acetaminophen, diclofenac, and their mixtures on river biofilm communities. Environ. Toxicol. Chem. 31:508-517 [doi:10.1002/etc.1723].

Leibold, M. A., and others. 2004. The metacommunity concept: a framework for multi-scale community ecology. Ecol. Lett. 7:601-613 [doi:10.1111/j.1461-0248.2004.00608.x].

Levi, P. S., J. L. Tank, J. Ruegg, D. J. Janetski, S. D. Tiegs, D. T. Chaloner, and G. A. Lamberti. 2013. Whole-stream metabolism responds to spawning Pacific Salmon in their native and introduced ranges. Ecosystems 16:269-283 [doi:10.1007/s10021-012-9613-4].

Likens, G. E., C. T. Driscoll, and D. C. Buso. 1996. Long-term effects of acid rain: Response and recovery of a forest ecosystem. Science 272:244-246 [doi:10.1126/science.272.5259.244].

Maddock, I. 1999. The importance of physical habitat assessment for evaluating river health. Freshw. Biol. 41:373-391 [doi:10.1046/j.1365-2427.1999.00437.x].

Mast, M. A., W. T. Foreman, and S. V. Skaates. 2007. Current-use pesticides and organochlorine compounds in precipitation and lake sediment from two high-elevation national parks in the Western United States. Arch. Environ. Contamin. Toxicol. 52(3):294-305 [doi:10.1007/s00244-006-0096-1].

Matthews, D. A., and S. W. Effler. 2006. Assessment of long-term trends in the oxygen resources of a recovering urban lake, Onondaga Lake, NY. Lake Reserv. Manage. 22:19-32 [doi:10.1080/07438140609353881].

McCormick, F. H., B. H. Hill, L. P. Parrish, and W. T. Willingham. 1994. Mining impacts on fish assemblages in the Eagle and Arkansas Rivers, Colorado. J. Freshw. Ecol.

9:175-179 [doi:10.1080/02705060.1994.9664884].
McGoff, E., and others. 2013. Assessing the relationship between the Lake Habitat Survey and littoral macroinvertebrate communities in European lakes. Ecol. Indic. 24:205-214 [doi:10.1016/j.ecolind.2012.09.018].
Meyer, J. L. 1997. Stream health: incorporating the human dimension to advance stream ecology. J. N. Am. Benthol. Soc. 16:439-447 [doi:10.2307/1468029].
Milhous, R. T., and T. J. Waddle. 2012. Physical Habitat Simulation (PHABSIM) Software for Windows (v.1.5.1). USGS Fort Collins Science Center.
Milly, P. C. D., K. A. Dunne, and A. V. Vecchia. 2005. Global pattern of trends in streamflow and water availability in a changing climate. Nature 438:347-350 [doi:10.1038/nature04312].
Moss, B., and other. 2003. The determination of ecological quality in shallow lakes: a tested expert system (ECOFRAME) for implementation of the European Water Framework Directive. Aquat. Conserv. Mar. and Freshw. Syst. 13:507-550 [doi:10.1002/aqc.592].
Murray, K. E., S. M. Thomas, and A. A. Bodour. 2010. Prioritizing research for trace pollutants and emerging contaminants in the freshwater environment. Environ. Pollut. 158:3462-3471 [doi:10.1016/j.envpol.2010.08.009].
Nakada, N., K. Kiri, H. Shinohara, A. Harada, K. Kuroda, S. Takizawa, and H. Takada. 2008. Evaluation of pharmaceuticals and personal care products as water-soluble molecular markers of sewage. Environ. Sci. Technol. 42:6347-6353 [doi:10.1021/es7030856].
Neilson, M. A., and others. 2003. Ecological monitoring for assessing the state of the nearshore and open waters of the Great Lakes. Environ. Monit. Assess. 88:103-117 [doi:10.1023/A:1025500619900].
Niemeijer, D. 2002. Developing indicators for environmental policy: data-driven and theory-driven approaches examined by example. Environ. Sci. Policy 5:91-103 [doi:10.1016/S1462-9011(02)00026-6].
Niyogi, D. K., W. M. Lewis, and D. M. McKnight. 2002. Effects of stress from mine drainage on diversity, biomass, and function of primary producers in mountain streams. Ecosystems 5:554-567 [doi:10.1007/s10021-002-0182-9].
Norris, R. H., and C. P. Hawkins. 2000. Monitoring river health. Hydrobiologia 435:5-17 [doi:10.1023/A:1004176507184].
Oviatt, C. A., A. A. Keller, P. A. Sampou, and L. L. Beatty. 1986. Patterns of productivity during eutrophication: a mesocosm experiment. Mar. Ecol. Progr. Ser. 28:69-80 [doi:10.3354/meps028069].
Palm, H. W., and S. Ruckert. 2009. A new approach to visualize ecosystem health by using parasites. Parasitol. Res. 105:539-553 [doi:10.1007/s00436-009-1423-z].
Pantus, F. J., and W. C. Dennison. 2005. Quantifying and evaluating ecosystem health: a case study from Moreton Bay. Aust. Environ. Manage. 36:757-771 [doi:10.1007/s00267-003-0110-6].
Parasiewicz, P. 2008. Habitat time-series analysis to define flow-augmentation strategy for the Quinebaug River, Connecticut and Massachusetts, USA. River Res. Appl. 24:439-452 [doi:10.1002/rra.1066].
———, and M. J. Dunbar. 2001. Physical habitat modelling for fish—a developing approach. Arch. Hydrobiol. Supp. 135/2-4:1-30.
Patil, G. P., R. P. Brooks, W. L. Myers, D. J. Rapport, and C. Taillie. 2001. Ecosystem health and its measurement at landscape scale: toward the next generation of quantitative assessments. Ecosyst. Health 7:307-316 [doi:10.1046/j.1526-0992.2001.01034.x].
Patrick, C. J., and C. M. Swan. 2011. Reconstructing the assembly of a stream-insect metacommunity. J. N. Am. Benthol. Soc. 30:259-272 [doi:10.1899/09-169.1].
Petrone, K. C., J. B. Jones, L. D. Hinzman, and R. D. Boone. 2006. Seasonal export of carbon, nitrogen, and major solutes from Alaskan catchments with discontinuous permafrost. J. Geophys. Res. Biogeosci. 111(G2) [doi:10.1029/2005JG000055].
Pfankuch, D. J. 1975. Stream reach inventory and channel stability evaluation. USDA Forest Service. Washington, D.C. RI-75-002.
Pietrock, M., and O. Hursky. 2011. Fish and ecosystem health as determined by parasite communities of lake whitefish (*Coregonus clupeaformis*) from Saskatchewan boreal lakes. Water Qual. Res. J. Can. 46:219-229 [doi:10.2166/wqrjc.2011.004].
Pilliod, D. S., C. S. Goldberg, R. S. Arkle, and L. P. Waits. 2013. Estimating occupancy and abundance of stream amphibians using environmental DNA from filtered water samples. Can. J. Fish. Aquat. Sci. 70:1123-1130 [doi:10.1139/cjfas-2013-0047].
Poff, N. L., and others. 1997. The natural flow regime: a new paradigm for riverine conservation and restoration. BioScience 47:769-784 [doi:10.2307/1313099].
———, J. D. Olden, N. K. M. Vieira, D. S. Finn, M. P. Simmons, and B. C. Kondratieff. 2006. Functional trait niches of North American lotic insects: traits-based ecological applications in light of phylogenetic relationships. J. N. Am. Benthol. Soc. 25:730-755 [doi:10.1899/0887-3593(2006)025[0730:FTNONA]2.0.CO;2].
Pont, D., R. M. Hughes, T. R. Whittier, and S. Schmutz. 2009. A predictive index of biotic integrity model for aquatic-vertebrate assemblages of Western US streams. Trans. Am. Fish. Soc. 138:292-305 [doi:10.1577/T07-277.1].
Quinn, J. M., N. R. Phillips, and S. M. Parkyn. 2007. Factors influencing retention of coarse particulate organic matter in streams. Earth Surf. Process. Landforms 32:1186-1203 [doi:10.1002/esp.1547].
Rankin, E. T. 1989. The qualitative habitat evaluation index (QHEI): rationale, methods, and application. State of Ohio, Environmental Protection Agency, Division of Water Quality. Columbus, Ohio.

Regier, H. A. 1993.The notion of natural and cultural integrity, pp. 3-18. *In* S. J. Woodley, J. J. Kay, and G. Francis (eds.), Ecological integrity and the management of ecosystems. St. Lucie Press.

Reuther, R. 1992. Arsenic introduced into a littoral freshwater model ecosystem. Sci. Total Environ. 115:219-237 [doi:10.1016/0048-9697(92)90331-L].

Roulet, N., and T. R. Moore. 2006. Environmental chemistry: Browning the waters. Nature 444:283-284 [doi:10.1038/444283a].

Ruaro, R., and E. A. Gubiani. 2013. A scientometric assessment of 30 years of the Index of Biotic Integrity in aquatic ecosystems: Applications and main flaws. Ecol. Indic. 29:105-110 [doi:10.1016/j.ecolind.2012.12.016].

Sabater, F., A. Butturini, E. Marti, I. Munoz, A. Romani, J. Wray, and S. Sabater. 2000. Effects of riparian vegetation removal on nutrient retention in a Mediterranean stream. J. N. Am. Benthol. Soc. 19:609-620 [doi:10.2307/1468120].

Sadro, S., and J. M. Melack. 2012. The effect of an extreme rain event on the biogeochemistry and ecosystem metabolism of an oligotrophic high-elevation lake. Arctic Antarc. Alpine Res. 44:222-231 [doi:10.1657/1938-4246-44.2.222].

Sanders, I. A., C. M. Heppell, J. A. Cotton, G. Wharton, A. G. Hildrew, E. J. Flowers, and M. Trimmer. 2007. Emission of methane from chalk streams has potential implications for agricultural practices. Freshw. Biol. 52:1176-1186 [doi:10.1111/j.1365-2427.2007.01745.x].

Sanderson, H., D. J. Johnson, T. Reitsma, R. A. Brain, C. J. Wilson, and K. R. Solomon. 2004. Ranking and prioritization of environmental risks of pharmaceuticals in surface waters. Regulat. Toxicol. Pharmacol. 39:158-183 [doi:10.1016/j.yrtph.2003.12.006].

Schaeffer, D. J., E. E. Henricks, and H. W. Kerster. 1988. Ecosystem health: 1. Measuring ecosystem health. Environ. Manage. 12:445–455 [doi:10.1007/BF01873258].

Scheffer, M., and S. R. Carpenter. 2003. Catastrophic regime shifts in ecosystems: linking theory to observation. Trends Ecol. Evol. 18:648-656 [doi:10.1016/j.tree.2003.09.002].

Schindler, D. W. 1977. Evolution of phosphorus limitation in lakes. Science 195:260-262 [doi:10.1126/science.195.4275.260].

———, K. H. Mills, D. F. Malley, D. L. Findlay, J. A. Shearer, I. J. Davies, and D. R. Cruikshank. 1985. Long-term ecosystem stress—the effects of years of experimental acidification on a small lake. Science 228:1395-1401 [doi:10.1126/science.228.4706.1395].

Schubert, S., A. Peter, R. Burki, R. Schoenenberger, M. J. F. Suter, H. Segner, and P. Burkhardt-Holm. 2008. Sensitivity of brown trout reproduction to long-term estrogenic exposure. Aquat. Toxicol. 90:65-72 [doi:10.1016/j.aquatox.2008.08.002].

Schwab, B. W., E. P. Hayes, J. M. Fiori, F. J. Mastrocco, N. M. Roden, D. Cragin,.and P. D. Anderson. 2005. Human pharmaceuticals in US surface waters: A human health risk assessment. Regulat. Toxicol. Pharmacol. 42:296-312 [doi:10.1016/j.yrtph.2005.05.005].

Schwarzenbach, R. P., B. I. Escher, K. Fenner, T. B. Hofstetter, C. A. Johnson, U. von Gunten, and B. Wehrli. 2006. The challenge of micropollutants in aquatic systems. Science 313:1072-1077 [doi:10.1126/science.1127291].

Shear, H., N. Stadler-Salt, P. Bertram, and P. Horvatin. 2003. The development and implementation of indicators of ecosystem health in the Great Lakes basin. Environ. Monit. Assess. 88:119-152 [doi:10.1023/A:1025504704879].

Smith, S. V., R. W. Buddemeier, F. Wulff, and D. P. Swaney. 2005. C, N, P fluxes in the coastal zone, pp. 95-143. *In* C. J. Crossland, H. H. Kremer, H. J. Lindeboom, J. I. Marshall-Crossland, and M. D. A. Le Tissier (eds.), Coastal fluxes in the anthropocene. Springer [doi:10.1007/3-540-27851-6_3].

Smith, V. H., G. D. Tilman, and J. C. Nekola. 1999. Eutrophication: impacts of excess nutrient inputs on freshwater, marine, and terrestrial ecosystems. Environ. Pollut. 100(1-3):179-196 [doi:10.1016/S0269-7491(99)00091-3].

Solomon, C. T., and others. 2013. Ecosystem respiration: Drivers of daily variability and background respiration in lakes around the globe. Limnol. Oceanogr. 58:849-866 [doi:10.4319/lo.2013.58.3.0849].

Soranno, P. A., K. S. Cheruvelil, K. E. Webster, M. T. Bremigan, T. Wagner, and C. A. Stow. 2010. Using landscape limnology to classify freshwater ecosystems for multi-ecosystem management and conservation. Bioscience 60:440-454 [doi:10.1525/bio.2010.60.6.8].

Soto-Galera, E., J. Paulo-Maya, E. Lopez-Lopez, J. A. Serna-Hernandez, and J. Lyons. 1999. Change in fish fauna as indication of aquatic ecosystem condition in Rio Grande de Morelia-Lago de Cuitzeo Basin, Mexico. Environ. Manage. 24:133-140 [doi:10.1007/s002679900221].

Staehr, P. A., and others. 2010. Lake metabolism and the diel oxygen technique: State of the science. Limnol. Oceanogr. Methods 8:628-644 [doi:10.4319/lom.2010.8.0628].

Stoddard, J. L., D. S. Jeffries, A. Lukewille, T. A. Clair, P. J. Dillon, C. T. Driscoll, and A. Wilander. 1999. Regional trends in aquatic recovery from acidification in North America and Europe. Nature 401:575-578 [doi:10.1038/44114].

———, and others. 2005. Environmental Monitoring and Assessment Program (EMAP): Western streams and rivers statistical summary. EPA 620/R-05/006, U.S. Environmental Protection Agency.

———, A. T. Herlihy, D. V. Peck, R. M. Hughes, T. R. Whittier, and E. Tarquinio. 2008. A process for creating multimetric indices for large-scale aquatic surveys. J. N. Am. Benthol. Soc. 27:878-891 [doi:10.1899/08-053.1].

Strange, E. M., K. D. Fausch, and A. P. Covich, 1999. Sustaining ecosystem services in human- dominated watersheds: Biohydrology and ecosystem processes in the South Platte River Basin. Environ. Manage. 24(1):39-54 [doi:10.1007/s002679900213].

Thom, R. M., and L. K. O'Rourke. 2005. Ecosystem health indicator metrics for the lower Columbia River and estuary partnership. Battelle Marine Sciences Laboratory.

Thomsen, P. E., J. Kielgast, L. L. Iversen, C. Wiuf, M. Rasmussen, M. T. P. Gilbert, L. Orlando, and E. Willerslev. 2012. Monitoring endangered freshwater biodiversity using environmental DNA. Mol. Ecol. 21:2565-2573 [doi:10.1111/j.1365-294X.2011.05418.x].

Thornton, T., and J. Leahy. 2012. Trust in citizen science research: A case study of the groundwater education through water evaluation and testing program. J. Am. Water Res. Assoc. 48:1032-1040 [doi:10.1111/j.1752-1688.2012.00670.x].

Tsai, A. Y., K. P. Chiang, J. Chang, and G. C. Gong. 2005. Seasonal diel variations of picoplankton and nanoplankton in a subtropical western Pacific coastal ecosystem. Limnol. Oceanogr. 50:1221-1231 [doi:10.4319/lo.2005.50.4.1221].

[USEPA] U.S. Environmental Protection Agency. 1989. Risk assessment guidance for Superfund Volume I: human health evaluation manual (part A). EPA 540-1-89-002. Office of Emergency and Remedial Response, US Environmental Protection Agency.

[USEPA] U.S. Environmental Protection Agency. 1990. Freshwater macroinvertebrate species list including tolerance values and functional feeding group designations for use in rapid bioassessment protocols. EA Report No.11075.05. U.S. Environmental Protection Agency.

[USEPA] U.S. Environmental Protection Agency. 1991. Technical support document for water quality based toxics control. EPA 505-2-90-001. Office of Water, US Environmental Protection Agency.

[USEPA] U.S. Environmental Protection Agency. 1998. National strategy for the development of regional nutrient criteria. EPA-8221-R-98-002. Office of Water, US Environmental Protection Agency.

[USEPA] U.S. Environmental Protection Agency. 2000a. Nutrient criteria technical guidance manual: lakes and reservoirs. EPA-822-B00-001. Office of Water, US Environmental Protection Agency.

[USEPA] U.S. Environmental Protection Agency. 2000b. Ambient water quality criteria recommendations. Information supporting the development of state and tribal nutrient criteria for lakes and reservoirs in nutrient ecoregion VI. EPA 822-B-00-008. Office of Water, US Environmental Protection Agency.

[USEPA] U.S. Environmental Protection Agency. 2005. National acid precipitation assessment program report to Congress: an integrated assessment. US Environmental Protection Agency.

[USEPA] U.S. Environmental Protection Agency. 2010. National lakes assessment: Technical appendix data analysis approach. EPA 841-R009-001a. Office of Water/Office of Research and Development.

[USEPA] U.S. Environmental Protection Agency. 2011a. 2012 National Lakes Assessment. Field Operations Manual. Office of Water. EPA 841-B-11-003. U.S. Environmental Protection Agency.

[USEPA] U.S. Environmental Protection Agency. 2011b. National Wetland Condition Assessment. Field Operations Manual. EPA 843-R-10-001.

[USEPA] U.S. Environmental Protection Agency. 2012. Water quality assessment and total maximum daily loads information (http://www.epa.gov/waters/ir/). Office of Water, US Environmental Protection Agency. <Web site accessed 25 June 2013>.

[USEPA]U.S. Environmental Protection Agency. 2013. National rivers and streams assessment 2008-2009 technical report draft. U.S. Environmental Protection Agency Office of Wetlands, Oceans, and Watersheds, Office of Research and Development.

[VANR] Vermont Agency of Natural Resources. 2008. Reach habitat assessments. Department of Environmental Conservation and Fish and Wildlife Department.

Vitousek, P. M., J. D. Aber, R. W. Howarth, G. E. Likens, P. A. Matson, D. W. Schindler, and D. Tilman. 1997. Human alteration of the global nitrogen cycle: Sources and consequences. Ecol. Appl. 7:737-750 [doi:10.2307/2269431].

Vugteveen, P., R. Leuven, M. A. J. Huijbregts, and H. J. R. Lenders. 2006. Redefinition and elaboration of river ecosystem health: perspective for river management. Hydrobiologia 565:289-308 [doi:10.1007/s10750-005-1920-8].

Wade, T. J., R. L. Calderon, E. Sams, M. Beach, K. P. Brenner, A. H. Williams, and A. P. Dufour. 2006. Rapidly measured indicators of recreational water quality are predictive of swimming-associated gastrointestional illness. Environ. Health Perspect. 114:24-28 [doi:10.1289/ehp.8273].

Wallace, J. B., J. W. Grubaugh, and M. R. Whiles. 1996. Biotic indices and stream ecosystem processes: results from an experimental study. Ecol. Appl. 6:140-151 [doi:10.2307/2269560].

Wang, L., J. Lyons, and P. Kanehl. 2003. Impacts of urban land cover on trout streams in Wisconsin and Minnesota. Trans. Am. Fish. Soc. 132:825-839 [doi:10.1577/T02-099].

Welsh, H. H., and G. R. Hodgson. 2008. Amphibians as metrics of critical biological thresholds in forested headwater streams of the Pacific Northwest, USA. Freshw. Biol. 53:1470-1488 [doi:10.1111/j.1365-2427.2008.01963.x].

Welter, J. R., S.G. Fisher, and N.B. Grimm. 2005. Nitrogen transport and retention in an arid land watershed: Influence of storm characteristics on terrestrial-aquatic linkages. Biogeochemistry 76(3):421-440 [doi:10.1007/s10533-005-6997-7].

Wetzel, R. G. 2001. Limnology: lake and river ecosystems. 3rd edition. Academic Press.

Wheaton, J., J. Brasington, S. E. Darby, A. Kasprak, D. Sear, and D. Vericat. 2013. Morphodynamic signatures of braiding mechanisms as expressed through change in sediment storage in a gravel-bed river. J. Geophys. Res. Earth Surf. 118 [doi:10.1002/jgrf.20060].

Wheaton, J. M., C. Gibbins, J. Wainwright, L. Larsen, and B.

McElroy. 2011. Preface: Multiscale feedbacks in ecogeomorphology. Geomorphology 126:265-268 [doi:10.1016/j.geomorph.2011.01.002].

Wiegner, T. N., S. P. Seitzinger, D. L. Breitburg, and J. G. Sanders. 2003. The effects of multiple stressors on the balance between autotrophic and heterotrophic processes in an estuarine system. Estuaries 26:352-364 [doi:10.1007/BF02695973].

Williamson, C. E., W. Dodds, T. K. Kratz, and M. A. Palmer. 2008. Lakes and streams as sentinels of environmental change in terrestrial and atmospheric processes. Front. Ecol. Environ. 6:247-254 [doi:10.1890/070140].

———, and others. 2014. Lakes as sensors in the landscape: Optical metrics as scalable sentinel responses to climate change. Limnol. Oceanogr. 59:840-850 [doi:10.4319/lo.2014.59.3.0840].

Wilcock, R. J., J. W. Nagels, G. B. McBride, K. J. Collier, B. T. Wilson, and B. A. Huser. 1998. Characterisation of lowland streams using a single-station diurnal curve analysis model with continuous monitoring data for dissolved oxygen and temperature. New Zeal. J. Mar. Freshw. Res. 32:67-79 [doi:10.1080/00288330.1998.9516806].

Wiley, M. J., L. L. Osbourne, and R. W. Larimore. 1990. Longitudinal structure of an agricultural prairie river system and its relationship to current stream ecosystem theory. Can. J. Fish. Aquat. Sci. 47:373-384 [doi:10.1139/f90-039].

Xu, F., Z. F. Yang, B. Chen, and Y. W. Zhao. 2013. Impact of submerged plants on ecosystem health of the plant-dominated Baiyangdian Lake, China. Ecol. Model. 252:167-175 [doi:10.1016/j.ecolmodel.2012.07.013].

Young, R. G., and A. D. Huryn. 1999. Effects of land use on stream metabolism and organic matter turnover. Ecol. Appl. 9:1359-1376 [doi:10.1890/1051-0761(1999)009[1359:EOLUOS]2.0.CO;2].

———, C. D. Matthaei, and C. R. Townsend. 2008. Organic matter breakdown and ecosystem metabolism: functional indicators for assessing ecosystem health. J. N. Am. Benthol. Soc. 27:605-625 [doi:10.1899/07-121.1].