Biological Assessment and Criteria Improve Total Maximum Daily Load Decision Making

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Abstract: Mandated total maximum daily load (TMDL) analyses present an excellent opportunity to restore the nation's degraded waters. The current norm for TMDL practice is, however, unlikely to achieve this goal without improved water quality standards plus systematic monitoring and assessment using biological criteria. Better than chemical and physical criteria alone, biological criteria link human actions, their impacts on water bodies, and societal goals, which are expressed as designated uses. To be adequate, monitoring should improve understanding of the connections among stressor, exposure, and response gradients. Water quality standards, monitoring, and assessment can improve water resources because they track water body condition, not the number of TMDLs completed. Federal and state leadership must set policy goals, as required by the Clean Water Act, and provide adequate fiscal and professional resources. States with high-quality programs should serve as models. Administrators should *use* the advances made in 2 decades of water resource science to improve their water management programs. Without such improvements, those involved in the TMDL process will continue to be frustrated, and the nation's waters will continue to decline.

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Introduction

In 1972 Congress defined a process to improve water quality, called total maximum daily load (TMDL). Total maximum daily load refers both to a pollutant's "total maximum daily load" that achieves compliance with a water quality standard and to the process of completing a specific plan that analyzes and allocates pollutant loads for specific water bodies or stream reaches. The TMDL goal is to restore impaired designated uses *and* protect waters that meet or exceed the criteria for such uses.

The components of a TMDL are to assess the quality of water bodies; to list impaired water bodies under Clean Water Act (CWA) section 303(d); to identify causes of degradation; to allocate pollutant loads to bring listed waters into compliance with CWA standards; and to manage to improve water quality, thereby removing waters from the 303(d) list. [Phrases like "303(d) list" or "305(b) report" are shorthand for mandates in the Clean Water Act.] "Completing TMDLs" refers to carrying out all these steps. A planning perspective centered on TMDL provides the best opportunity in a generation to improve the nation's water resources.

Nevertheless, federal and state implementation of the TMDL mandate languished for 20 years before a report [General Accounting Office (GAO) 1989] called attention to this shortfall, and a spate of judicial decisions required the U.S. Environmental Protection Agency (EPA) and the states to complete about 21,000 TMDL analyses (Houck 1999). According to the National Academy of Sciences [National Research Council (NRC) 2001], the TMDL approach advocated by EPA and implemented by many states is deeply flawed, from how it lists impaired water bodies to how it diagnoses causes of degradation, to how it makes decisions about restoring damaged water bodies.

Weaknesses include the basis for listing many "impaired" waters, failures to list truly impaired waters, limited efforts to determine whether "completing" a TMDL actually improves a water body, a zeal to delist that diverts attention from the need to improve the listing process itself, and completing TMDLs with little or no ground-truthing. Most TMDLs are based on inadequately validated mechanistic simulation models.

Another flaw is a tendency to ignore or underrate important degradation agents (altered flows, changes in physical habitat, presence of alien taxa such as noxious plants, and so on; NRC 2001). Total maximum daily load allocations emphasize a small set of individual pollutants (the top five are sediment, pathogens, metals, nutrients, and organic enrichment; GAO 2003b) while many serious pollutants and other forms of pollution—not to mention important interactions among them—go unexamined. Pollutants are substances added to waters by human activity [CWA section 502(6)]. The Clean Water Act further defines pollution as human-induced alteration of waters caused by pollutants as well as nonpollutant agents, such as flow alteration, loss of riparian zone, physical habitat alteration, and introduction of alien taxa [CWA section 502(19)].

Beyond many models' comparatively simple assumptions, many TMDLs do not incorporate important interactions between nonpollutant influences and pollutants. A more complete assessment process by Ohio EPA (1997), for instance, identifies altered physical habitat and altered river flows along with siltation, organic enrichment, and nutrients as the top five causes of impairment. But because of TMDLs' emphasis on chemistry and toxi-

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cology, nonpollutant forms of pollution (or degradation) are rated inordinately low or ignored in most states, despite their prominence in national assessments of aquatic resources (Judy et al. 1984; NRC 1992; Wilcove and Bean 1994).

Furthermore, inadequately defining designated uses and delineating impaired waters undermine the credibility of impaired segment listings. Utah, for example, assigned designated uses over a 4–5 day period using best professional judgment because of concerns that grant funds would be withheld (GAO 2003b). Waters listed as ‘‘potentially impacted’’ by nonpoint sources for CWA section 319 funding purposes in several states were later ‘‘transferred’’ to the 303(d) list merely because they appeared in the 305(b) report. Listings driven by the search for funding rather than actual water body assessments are common under new TMDL drivers [section 303(d)]. For five states, more than 50% of water bodies need changes in designated uses; for 28 states, 1–20% of water bodies need such changes (GAO 2003b). Deficiencies in designating water bodies must be resolved before additional TMDL shortcomings can be addressed. Finally, the actual condition of water bodies is often poorly known, and the effectiveness of existing TMDL activities is not routinely evaluated.

Is the TMDL process so fundamentally flawed that it should be abandoned, or can it be improved to restore the quality of the nation’s waters? We believe a credible fix requires EPA, the states, and stakeholders to view water resources in new ways. Specifically, scientists, managers, policymakers, environmentalists, and regulated communities must work together to implement a seamless and integrative approach to water resource management—as charged in the Clean Water Act, restore and maintain the biological integrity of the nation’s waters.

The four primary points of this article are:

1. As currently conceived and implemented, the TMDL program is seriously flawed;
2. Effective TMDL programs require rigorous water quality standards (including refined designated uses and biological criteria) and rigorous monitoring and assessment programs;
3. Research programs of the last 2 decades have produced the tools needed to complete scientifically and statistically rigorous TMDLs; and
4. Agency leadership should embrace those programs to ensure success in attaining Clean Water Act goals.

Total Maximum Daily Loads Require Clear Goals and Measures of Water Body Condition

Two crucial elements of successful TMDL programs are: (1) water quality standards and (2) monitoring and assessment (NRC 2001). Water quality standards (WQS) include designated uses (water-body specific goals), criteria (numeric and narrative benchmarks), and an antidegradation policy. Criteria are design targets to restore and protect designated uses and assessment thresholds for determining status. Impaired water bodies are included in a state’s 303(d) list; each must be the subject of a TMDL.

Monitoring measures chemical, physical, and biological factors, and assessment evaluates the condition of places to determine if applicable criteria and designated uses are being attained (Karr and Chu 1999). Antidegradation and the concept of existing use are intended to maintain conditions present at the time federal water quality regulations were adopted.

The clarity of water quality standards, especially the specification and appropriateness of designated uses, and actual monitoring and assessment determine if impaired water bodies are correctly identified. Inadequate WQS may permit resource degradation, and inadequate monitoring and assessment may lead to inaccurate description of water body condition. The latter leads to incomplete or inaccurate 303(d) lists and unnecessary expenditures on unneeded TMDLs or failure to develop needed TMDLs.

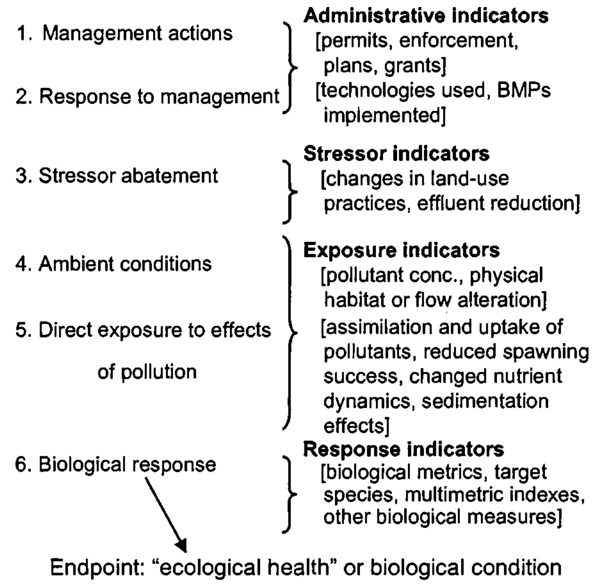

Fig. 1. Hierarchy of monitoring and assessment indicators. All can be used to measure and manage environmental progress, but only biological responses focus on end outcomes (modified from U.S. Environmental Protection Agency 1995b and Yoder and Rankin 1998).

Bioassessments more accurately detect and quantify aquatic life impairments than do pollutant-specific criteria and sampling (Karr et al. 1985; Rankin and Yoder 1990). Biocriteria-based assessments commonly detect 100% more impairment than do chemical assessments. Even when biological and chemical assessments agree, the information communicated by biology is more complete because it identifies stressors and the results of processes not completely revealed by chemical assessments (Rankin and Yoder 1990). And to be effective, TMDL programs must identify stressors accurately.

We do not advocate abandoning chemical assessments; rather, we note the inadequacy of assessments that do not recognize the hierarchy of indicators in Fig. 1. From counts of management actions to measures of biological response, this hierarchy provides a framework for understanding, regulating, and managing impacts to surface waters (U.S. EPA 1995b). It strengthens our understanding of the interrelations between human actions and water body condition. Unfortunately, this framework, initially developed by EPA in the late 1980s, is not widely employed in the United States.

Water Quality Standards

So-called designated uses are supported by numeric or narrative criteria intended to protect a given use, such as drinking, swimming, or aquatic life. Carefully formulated designated uses couple a descriptive narrative with supporting numeric criteria, including how they were derived and calibrated. Unfortunately, designated uses in many states lack the narrative or numeric specificity to account for regional and local properties or the ecological potential of specific water bodies (GAO 2003b). Although water quality standards offer a plausible structure to protect and restore

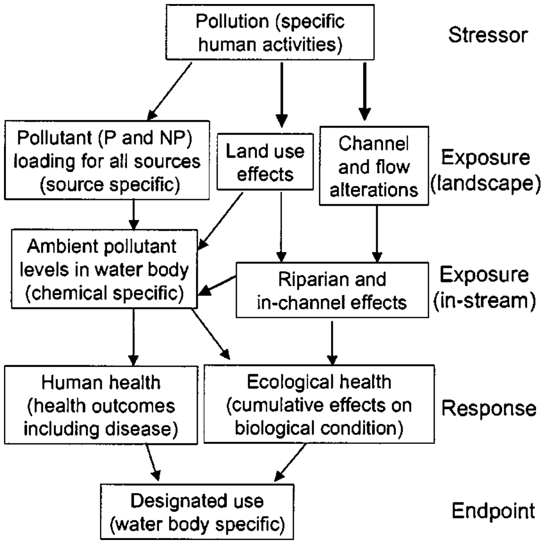

Fig. 2. Position of criterion (stressor, exposure, or response), illustrating relationships among human activities, specific types of criteria, and designated uses that define endpoints of interest to society (modified from National Research Council 2001)

water bodies, vaguely defined designated uses (e.g., ''aquatic life support'' or ''cold water fish assemblage'') do not. Vaguely phrased general uses do not consider differences in stream size, gradient, topography, climate, or other factors. They do not account for the cumulative effects of multiple stressors and exposures that limit aquatic life. They lead to a one-size-fits-all TMDL process.

A more reasoned approach employs tiered aquatic life uses in conjunction with a biological condition gradient to specify the ecological potential of each water body, thereby leading to more cost-effective decisions for pollution control and restoration. In setting designated uses for urban or agricultural landscapes, for example, expectations might be different from those in parks and reserves. Without tiered aquatic life uses, real differences among waters are not accounted for in the listing of impaired water bodies, an oversight that can lead to inappropriate TMDL decisions.

Another common weakness of WQS is emphasis on a few select pollutants as surrogates of aquatic life use attainment (Karr and Dudley 1981; Karr 1991). Misapplied surrogates are pervasive in 305(b) reporting and TMDL analyses, representing the substitution of stress and exposure indicators for response indicators (Yoder and Rankin 1998).

The NRC (2001) characterized the crux of indicator selection as the ''position of the criterion'' relative to the designated use (Fig. 2), stressing the importance of connecting biological criteria directly to explicit aquatic life (designated) uses. Although sometimes easier to develop and measure, criteria more removed from designated uses (e.g., administrative or chemical measures) are less accurate surrogates (NRC 2001). Criteria positioned closer to biological condition are more integrative and accurate indicators that explicitly defined uses have been attained. Conversely, attainment of designated uses is more reliable, accurate, and scientifically rigorous when ascertained by refined aquatic life uses and biocriteria.

The NRC is not the only organization to recognize problems associated with surrogate performance measures. The U.S. General Accounting Office (GAO 2003a) notes that dependence on administrative performance measures (e.g., number of environmental standards established, permits issued, and enforcement actions taken, all referred to as outputs) still limits program effectiveness, including EPA's ability to assess risk; in 1999, for example, 86% of 278 EPA performance measures were outputs rather than end outcomes (measures that directly measure environmental conditions). The percentage of environmental performance measures increased from 7% in 1999 to 27% in 2003. But even so, most end outcomes in GAO's analysis are chemical, not biological (the gold standard endpoint).

Another general TMDL program weakness is the lack of information to support impaired water delineations [303(d) lists] or management decisions in TMDLs. Yet collecting more chemical data—for example, NRC's (2001) call for more chemical water quality samples to address statistical issues—cannot address three important problems. First, chemical criteria do not document biological effects of pollution, or even if there are any. Second, if chemical criteria are not exceeded, a healthy biota is inappropriately assumed. Third, collecting chemical samples cannot identify the effects of chemical contaminants not included in conventional analyses (e.g., pharmaceuticals and other contaminants; Kolpin et al. 2002).

Chemical assessments more likely lead to interpretation errors than do biological assessments, for example listing of water bodies as impaired when they are not (type I error) or failing to list when they are impaired (type II error). Type I error often garners more attention (NRC 2001) because stakeholder groups, especially regulated entities, implicated by erroneously identified impairments can rightly claim ''bad science.'' A preoccupation with this issue can detract from more important phenomena involved in type II errors. Studies using empirical data (Rankin and Yoder 1990; D. Drake, Oregon Department of Environmental Quality, personal communication, 2002) show that type II errors are in fact the most common, leaving impairment undiagnosed and management agencies unaware.

Common sources of type I and II errors include improper stratification of ecological potential across regional landscapes (e.g., not incorporating the influence of stream size) or poorly developed or improperly selected indicators of water body condition. For aquatic life uses, assessments may reach completely different conclusions if based on chemical sampling versus biological assessments. Poorly conceived and implemented biological indicators can lead to similar errors.

Generally, however, biological assessments can overcome statistical difficulties posed by chemical sampling, avoid the cost of frequent chemical sampling, and measure actual resource condition rather than depending on chemical surrogates of biological response. An improved approach links activities like wastewater treatment directly to biological responses, thereby directing management toward practices that protect explicitly stated designated uses, as opposed to seeking some ''correct'' chemical pollutant threshold.

Monitoring and Assessment

Monitoring and assessment provide the information needed to determine if a water body is impaired and if it responds to TMDL implementation (GAO 2000; NRC 2001). Monitoring has not been a priority for federal and state water quality managers; only 0.2% of the resources devoted to the management of water quality in the United States were used for ambient monitoring in the early

1990s [Intergovernmental Task Force on Monitoring Water Quality (ITFM) 1992]. Despite calls from scholars (Karr and Dudley 1981; Van Putten 1989; Keeler and McLemore 1996) and congressional staffers (Anonymous 1981); the example of leading state programs (Ohio EPA 1987; 1989a,b; Courtemanch 1995; McCarron and Frydenborg 1997); and recent federal initiatives to foster results-driven approaches (ITFM 1992, 1995; U.S. EPA 1995a,b) , water quality management programs still emphasize administrative measures, or outputs, instead of biological responses (see Fig. 1), an end outcome (GAO 2003a).

The EPA and state programs emphasize policies based on simplistic assumptions and paradigms; monitoring is viewed as a tangential ‘‘extra.’’ When the TMDL process considers a few pollutants without recognizing that degradation derives from more than elevated concentrations of chemicals, it defies common sense even as it ignores the lessons of three decades of research and application (Karr 1991; ITFM 1992, 1995; Yoder 1998).

Evidence that management programs must deal with pollutants and pollution is abundant (Karr and Dudley 1981; Yoder and Rankin 1998; Karr and Chu 1999). Unfortunately, and despite calls for this approach for 20 years (U.S. EPA 1984, 1988, 1990; ITFM 1992, 1995), this concept has not been fully embraced by those charged with restoring and maintaining water bodies.

Historic neglect of the strategic aspects of monitoring has unintended consequences. First, lack of quality data limits our ability to accurately define impaired waters causing erroneous listings and failure to list impaired water bodies. Second, data adequate to diagnose causes of impairment, a crucial step in the development and implementation of TMDLs, are not available. Third, the lack of commitment to monitoring seriously impedes refinement of TMDL programs, including adapting them to regional contexts and emerging problems. Agency professionalism has not kept pace with evolving water resource challenges. Despite the availability of better diagnostic tools (Karr et al. 1986; U.S. EPA 2000; Morley and Karr 2002; Yoder and DeShon 2003), agency diagnoses of the causes of degradation are not as sophisticated as they could be. Agencies have failed to ‘‘learn by doing,’’ which would have conferred practical benefits exceeding those from conventional research.

Fourth, management success is limited because agencies have not shifted to data- and results-driven approaches, have not shifted from outputs to end outcomes. These deficiencies can be overcome with high-quality biological assessments, the ‘‘raw material’’ for improving TMDLs, combined with more-focused and more-detailed chemical, physical, and biological criteria in water quality standards.

Still EPA and some states are improving their programs. The EPA development of tiered aquatic life uses is an important step. A state agency that adapted seamlessly to TMDLs is the Ohio EPA, which sustains a monitoring and assessment program supported by sufficiently detailed designated uses, criteria, and routine use attainability analysis (Ohio EPA 1999b; U.S. EPA 2002a).

Finally, an improved TMDL approach (NRC 2001) would break the logjam of current listings and focus energies on priority TMDL needs. Many 303(d) listings of waters as impaired are not backed by comprehensive statements of designated uses or data on water body condition. The NRC recommends that states first put these water bodies on a ‘‘preliminary’’ list until properly defined standards and data support those listings. Water bodies should be moved to an action list or removed from the 303(d) list altogether when acceptable data become available.

Key Actions to Improve Total Maximum Daily Load Process

Beyond improving designated uses and monitoring and assessment programs, managers can take other actions to produce more effective TMDLs. The length and complexity of a state’s 303(d) list, for example, has become *the* primary agency concern. Agency responses emphasize completion of TMDLs despite the fact that many listings were made ‘‘without *any* supporting water quality data,’’ and some impaired waters ‘‘have yet to be identified and listed’’ (NRC 2001). Moreover, the completed TMDLs often ignore substantial causes of water resource degradation. Here we propose specific actions to improve the TMDL process.

Restore Biological Focus

Before World War I, water managers in Europe and North America recognized the connection between pollution, river biology, and human health (Davis 1995). Changes in aquatic biota were the primary indicators of humans’ effects. As industrial activity expanded, water programs aimed to control chemical pollutants instead. Mass balance, pollutant loadings, and dilution were central concepts, and treatment technology was emphasized. The discharge of waste was seen as a ‘‘universal practice and ... a legitimate use of stream waters’’ (Evans 1965). The capacity of a water body to assimilate pollutants was a central focus, because ‘‘the various forms of life in a river are purely incidental, compared with the main task of a river, which is to conduct water runoff from an area toward the ocean’’ (Einstein 1972). Government programs of the 1960s and 1970s were grounded in these incomplete views.

A return to biological indicators and concepts occurred later in the 20th century, with two important implications for the TMDL process. First, nonpoint-source pollutants involve constituents, processes, and interactions not directly amenable to mass-balance pollutant approaches. Recent work in headwater streams in Ohio, for example, documents interactions between physical habitat and nonpoint pollutants like phosphorus (Ohio EPA 1999a). Pollutant impacts are reduced by headwater streams with intact riparian zones and natural channel morphology. Excess nutrients are sequestered in biomass; channel shading reduces sunlight (a principal limiting factor in algal production). In contrast, nuisance algal blooms degrade streams with degraded riparian zones. Management aimed solely at reducing pollutant runoff ignores interactions vital to assimilation of excess nutrients.

Managers unfamiliar with these interactions cannot diagnose or treat such impaired water bodies. Yet EPA has proposed categorizing impairments according to simple, single-factor causes (e.g., nitrogen alone when multiple factors may be involved at a site). EPA’s approach reinforces the flawed pollutant-only approach, instead of recognizing the mosaic of factors and their interactions responsible for water body impairment.

Second, an integrative view of stressors combined with a focus on end outcomes (ideally, biological endpoints) will improve TMDLs. Permit terms and conditions for a sewage treatment plant, for example, prescribe actions on the basis of the whole treatment process [e.g., best available demonstrated control technology (BADCT)]. This approach to advanced treatment is replacing secondary treatment as the norm in many places, a shift that should be accompanied by better diagnosis of the causes of degradation. Implementing standard practices, such as minimum riparian buffer widths or the general application of improved land use practices, will improve water resources even as it decreases the backlog of river reach TMDLs. Another advantage of a biological focus is the supporting evidence that comes from the accumulation of better data.

The Auglaize River in Ohio illustrates the power of integrating biological and other data. Before 1990, few farmers implemented "best management practices," such as conservation tillage or no tillage (<5% of cultivated acres). An intensive program to change tillage practices began in 1990; most farmed acres were treated by 1993 (Fig. 3, upper panel). According to data collected by U.S. Geological Survey (USGS), annual loadings of total suspended solids (TSS) in the river decreased by 50% in the late 1990s (Fig. 3, middle panel; Myers et al. 2000). The 60% decrease in summer TSS was even more striking; high summer TSS, when flows are generally low, are probably due to excessive algal growth rather than suspended mineral soils. Most important perhaps, these changes in land practices were followed by improved biological condition, as assessed by the fish index of biological integrity (IBI) (Fig. 3 lower panel). Median IBI increased by about 15 points, from the high 30s (fair) to the low 50s (exceptional), after 1 decade of conservation tillage on most of the basin's cultivated lands. Changes in fish assemblages captured in IBI metrics were expected with reduction in nonpoint stressors such as suspended solids, nutrients, and their primary (e.g., substrate quality) and secondary (e.g., diel dissolved oxygen regime) effects. As a result, the river came into compliance with Ohio's biological criteria for "warmwater" and even attained an "exceptional warmwater" rating (as defined in Ohio Administrative Code 3745-1-07).

This example illustrates five important points. First, systematic, standardized monitoring strengthens our ability to connect stressor, exposure, and response gradients (see Fig. 2). The Auglaize also illustrates the role of systematic monitoring in characterizing resource condition for present and future management. Many existing programs, especially national nonpoint-source programs, lack an integrative approach to indicators or the spatial and temporal detail to document program effectiveness.

Second, biological data represent the gold standard—the endpoint outcome of concern—and thus constitute a powerful complement to chemical and physical data. Third, when chemical and biological data are combined, fewer chemical data are required to demonstrate significant progress, both statistically and biologically, toward attainment of designated uses. Fourth, discerning the biological benefits of improved agricultural practices can be simple, fast, and direct. Restoring streams and rivers impaired by nonpoint sources is possible with reasonable changes in agricultural practices. The substantial biotic improvement over a broad area is superior to the more common segment-by-segment, pollutant-specific TMDL approach. Tillage practices in this example are a nonpoint analog of the generic requirement for secondary treatment (and now BADCT) in point-source control that was so instrumental in improving water bodies throughout Ohio. Finally, the Auglaize River illustrates a remarkable degree of biological improvement after the relatively simple task of improving land management practices.

Although the numeric drivers of permit limitations are historically based on individual pollutant criteria, the links between wastewater treatment and the receiving waterbody are far more complicated than the univariate relationships that individual pollutant-by-pollutant approaches imply. In reality, the nonlinear interactions of all stressors (multiple pollutants and other forms of pollution) are captured by an integrative focus on biological responses and causal analysis that employs chemical, physical, and biological indicators.

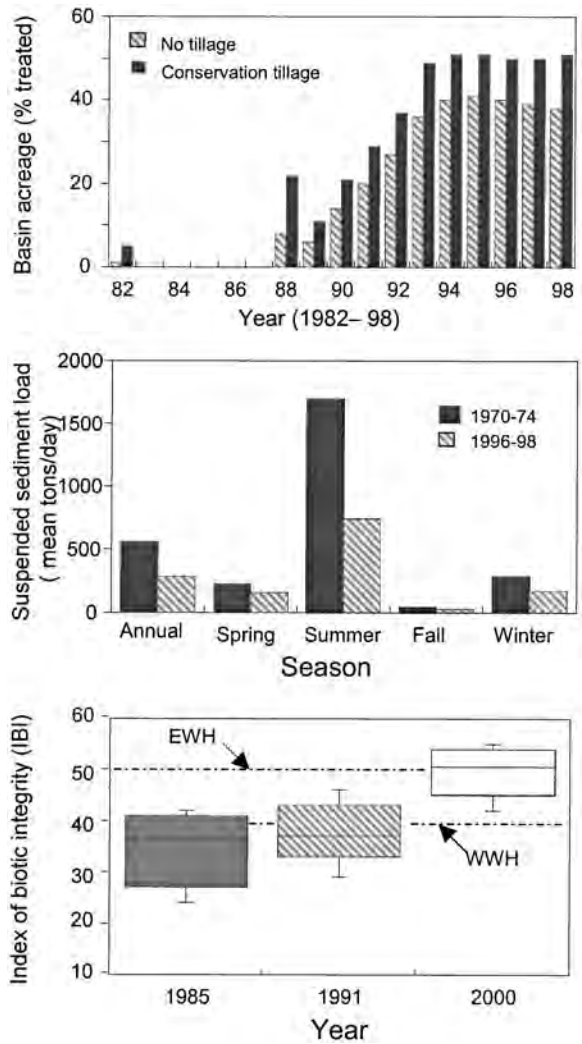

Fig. 3. Connections between land management changes and improvements in biological condition in Auglaize River mainstem, Ohio. (Upper Panel) Percentages of regional farmed acreage managed with conservation tillage or no tillage best management practices increased with incentives for their use, beginning in 1990 (data from Myers et al. 2000). (Middle Panel) Loading of suspended solids declined sharply in 1990s, especially in summer, at U.S. Geological Survey Auglaize River gaging site, Ft. Jennings, Ohio, after regional adoption of conservation and no tillage (data from Myers et al. 2000). (Lower Panel) River health, as measured by fish index of biotic integrity at multiple sampling stations between river miles 39 and 80, generally did not attain minimum biological criterion level for warmwater habitat (index of biotic integrity=40) in 1970s and 1980s. After widespread adoption of conservation tillage and no tillage, all sampling stations came into compliance with warmwater habitat criteria (index of biotic integrity⩾40), and many sites attained index of biotic integrity levels characteristic of exceptional warmwater habitat (index of biotic integrity⩾50; data from Ohio Environmental Protection Agency).

Link Stressors to Gradients of Biological Response

Awareness of biological response was central to early efforts to protect water quality. Early in the 20th century, the Saprobien system recognized two facts ignored by later emphasis on chemical criteria: not all water bodies are the same (large or small streams; cold, cool, or warmwater streams), and human effects

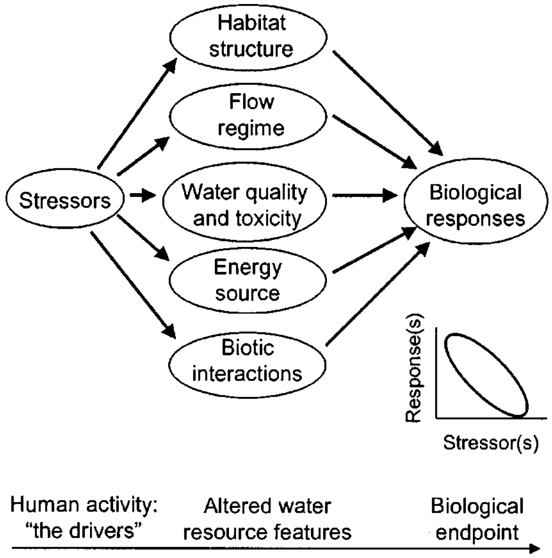

Fig. 4. Linkages from human activity (stressors or drivers of system change) through five major water resource features altered by human activity, to biological responses producing ambient condition, i.e., biological endpoints of primary interest in biological assessment programs. Model illustrates multiple causes of water resource changes associated with human activities. Inset illustrates relationship between stressor dose and gradient of biological responses that signal good biological metric.

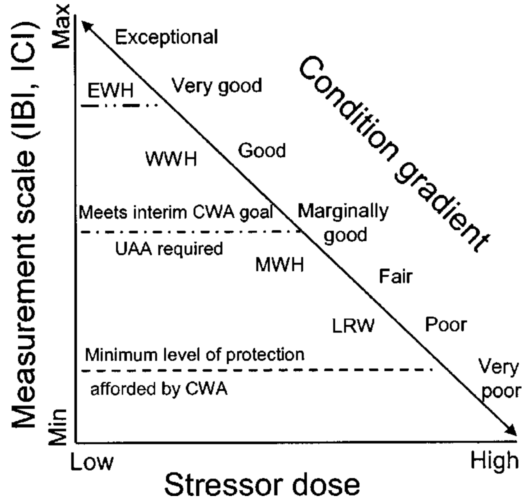

Fig. 5. Relationship between stressor dose and biological measurement scale, such as index of biotic integrity or invertebrate community index, showing level of biological condition exceptional to very poor) and associated aquatic life designated uses—(exceptional warmwater habitat, warmwater habitat, modified warmwater habitat, and limited resources waters—as defined by Ohio Environmental Protection Agency and as codified in Ohio water quality standards.

inevitably lead to a continuum, or gradient of biological condition. One-size-fits-all chemical criteria, without a biological complement, produced a two-class assessment hierarchy, variously termed impaired or unimpaired, in compliance or not. Simple to understand and easy to use, this dichotomy is neither sufficiently accurate nor robust enough to address the issues that the TMDL process was crafted to resolve (Karr and Chu 1999).

Recent advances in biological assessment open windows of understanding beyond the pass–fail paradigm. First, modern analytical tools, such as multimetric biological indexes, enhance our ability to measure quality in a manner that communicates the severity and extent of impairment. They also allow us to recognize and interpret patterns in biological data, including the relationships among chemical, physical, and biological stressors and their influences on the five features of water resources altered by human activity (Fig. 4) and along biological condition gradients (Fig. 5).

Second, combining the biological condition gradient with pattern recognition as a biota changes enables refinement of aquatic life uses. Biological condition gradients make it easier to communicate assessment results in numbers and words to the public and policymakers and in legal and regulatory proceedings, where confidence intervals or hypothesis testing are required. Five to six nonoverlapping categories of biological condition can be distinguished whether one studies fish (Fore et al. 1994) or invertebrates (Doberstein et al. 2000).

The multiple dimensions implicit in multimetric biological indexes are amenable to rigorous statistical analyses (e.g., Fore et al. 1994; Hughes et al. 1998; Doberstein et al. 2000; Bryce et al. 2002; Fore 2002; Mebane et al. 2003), providing TMDLs with better data and thus making them more relevant to common water resource challenges. Multiple biological dimensions linked to human actions and to physical and biological conditions lead to more rigorously defined management goals and more effective decision making. The concept of a gradient of biological condition (as opposed to the regulatory dichotomies of impaired versus unimpaired) is revolutionizing water quality analysis, because it integrates monitoring with the concept of tiered designated uses. Another important outcome is refined thresholds of chemical and physical stressors. Rigorous biological assessments provide more effective and focused diagnostic capability (Norton et al. 2000; Yoder and DeShon 2003), diagnostic power that is assumed but not present in most uses of chemical data.

Finally, reemergence of integrative biological approaches connects water programs to the broader biological mandate of the CWA (Karr 1991; Karr and Chu 1999, 2000). These innovations have revolutionized some state programs [Ohio (Ohio EPA 1987; Yoder and Rankin 1998); Maine (Davies et al. 1995; Courtemanch 1995); Florida (McCarron and Frydenborg 1997); Idaho, Idaho Department of Environmental Quality (IDEQ) 2003] and are making inroads in others (U.S. EPA 2002a). Full realization of the potential of this approach, however, will happen only if EPA makes biological criteria a program priority, something it has not done, despite adopting national program guidance in 1990 (U.S. EPA 1990).

Align Federal and State Policies and Leadership with Scientific Advances

Although exceptions exist, the management of EPA and many state agencies continue to resist the scientific advances of the last 2 decades, advances that can, if used properly, better focus regulatory and incentive programs, save money, and improve the quality of water resources (NRC 2001). This resistance began 30 years ago when the EPA administrator testified against adoption of the "integrity" goal in the 1972 Amendments to the Water Pollution Control Act (Committee on Public Works 1973). Fortunately, others within EPA fostered important advances in academia, national labs, and state programs (including much of the authors' work in the last 30 years). More recent examples abound from programs such as EPA's Environmental Monitoring and Assessment Program, other federal agencies, several states, and academic scientists; ongoing studies focus on large rivers, wetland, coastal, and coral reef environments. The benefits of these efforts are well documented in publications cited throughout this paper.

Yet too many federal and state programs have been slow to implement these advances. Equally important, state and federal research efforts are seldom integrated with regulation and policy. Sometimes federal and state agencies do not even recognize that their respective goals and approaches do not match. These problems are made worse by lack of coordination and communication. Too often, entrenched bureaucracies and lobbying groups advocate the status quo, pushing for ''cheaper and easier'' methods while overstating their capabilities.

In essence, incorporating the conceptual frameworks and scientific accomplishments of the past two decades into ongoing TMDL programs is crucial, even more so than funding additional research. A substantial body of scientific and management information is available to improve TMDLs. The scientific community has time and again carefully evaluated and validated that knowledge, but it is still not being used as it should be. We urge federal and state agency leaders to change this situation.

Align Quantitative Models with Ecological Concepts

Conventional TMDL programs use modeling exercises of the following form (NRC 2001) to define pollutant loads, usually in pounds per day

$$\mathrm{TMDL} = \mathrm{WLA} + \mathrm{LA} + \mathrm{MOS} + \mathrm{BG}$$

where WLA=waste–load allocation from defined point sources; LA=load allocation from nonpoint sources; MOS=margin of safety to ensure attainment of water quality standards; and BG =background.

The formula excludes all forms of pollution not defined as pollutants. It could be argued that the margin of safety term covers components of those excluded factors. But this argument is both indirect and scientifically indefensible, because important factors responsible for degradation are missed; interactions among pollutants or among pollutants and other forms of pollution are also ignored.

One advance of the past 2 decades is recognizing the diverse human activities that alter water resources and the extent to which those activities interact with topographical, geological, climatological, and biological differences among watersheds (Karr and Rossano 2001). One can identify five features of water resources that the cumulative effects of human activity alter, with consequences for aquatic biota (Karr et al. 1986; Karr 1991):

1. Energy source: changes in the food web, including nutrients, organic material inputs, seasonal cycles, primary and secondary production, and sunlight;
2. Chemical variables: changes in chemical water quality, including DO, pH, turbidity, hardness, alkalinity, solubilities, adsorption, nutrients, organics, toxic substances, temperature, and sediment;
3. Flow regime: modification of flows, including precipitation, seasonal pattern, land use, runoff, velocity, groundwater, and flow extremes;
4. Habitat structure: alteration of physical habitat, including bank stability, current, gradient, instream cover, vegetative canopy, substrate, current, sinuosity, width, depth, pool-to-riffle ratios, riparian vegetation, sedimentation, and channel morphology; and
5. Biotic factors: changes in biotic interactions, including alien taxa, feeding, reproduction, predation, overharvest by sport, commercial, and subsistence fishers, diseases, parasitism, and competition.

When human actions influence one or more of these factors, or their interactions, the aquatic biota changes (Fig. 4). The severity and degree of the biological response to these impacts are ultimately what is important, not the mere presence of an impact. Understanding these interactions ought to guide selection of indicators for monitoring programs (Karr 1991; Yoder 1998).

A more realistic approach to TMDL modeling than the conventional one might adopt the following conception:

$$\mathrm{UA} = f(\mathrm{PL},\mathrm{FR},\mathrm{ES},\mathrm{HS},\mathrm{BI})$$

where UA=designated use attainment as an integrative function of the effects of human activity; PL=pollutant loads, including point and nonpoint; FR=flow regime or hydrological changes; ES=energy source; HS=physical habitat structure; and BI =biotic interactions.

These five factors are not additive, allowing for both univariate effects and multifactor interactions (linear and nonlinear). When water resource evaluations are integrated with detailed chemical, physical, and biological indicators and knowledge of their interactions, this equation can guide the discovery of the causes of degradation, even though it may not be possible to specify all model details or to parametrize all components. Without the breadth of this conceptual model, the TMDL approach is incomplete, yielding inadequate conclusions and inappropriate management strategies.

These suggestions go well beyond the use of site-specific variables such as hardness and pH as modifiers for WQS. They include tiered use designations and variable thresholds for determining both impairment and incremental changes in quality. The model can also facilitate understanding of connections between human actions and environmental results.

Efforts to align water quality models with ecological reality should also be integrated with policy initiatives. These diverse programmatic activities should not be seen as independent or disconnected as they have been in the past. In a recent report, ''The Twenty-Needs Report: How Research Can Improve the TMDL Program,'' (U.S. EPA 2002b), EPA responds to reviews of TMDL and other water programs (U.S. EPA 1998; GAO 2000, 2003a,b; NRC 2001). Unfortunately, programmatic isolation is obvious in the ''Twenty-Needs'' report's executive summary: ''This document does not represent or modify EPA's TMDL program policy or guidance and is limited to analysis and recommendations concerning scientific issues.'' This disclaimer does little to reinforce our confidence that connections among researchers, managers, practitioners, and policymakers will be strengthened as a result of comprehensive program reviews (U.S. EPA 1998; GAO 2000, 2003a,b; NRC 2001) or that existing knowledge will be effectively applied.

The ''Twenty-Needs'' report, like so many documents generated by or for EPA, is about science and research as recently conducted *within* EPA. Of 26 items cited as references, 14 are EPA publications; five more are by EPA staff. Apparently hundreds of papers and a dozen books documenting recent scientific and management advances were overlooked or ignored by EPA authors. Calls for more research that neither recognize nor use existing research knowledge inevitably sustain the status quo. Furthermore, the report does not recognize the questionable foundation of many TMDL analyses. Even broader issues that are discussed by us here are framed in a research sense rather than as an urgent priority for implementation in state programs.

Uncertainties persist even when conceptual models like the one advocated here are used. But those uncertainties are small when compared with the uncertainties created by excluding important stressors from TMDL analyses or by ignoring biological endpoints as the best reflection of whether designated uses have been attained.

Improve Monitoring and Assessment

Monitoring has long been viewed as an analog of the radar ''gun'' used to identify speeders: it detects when environmental criteria are exceeded. Although important for that purpose, monitoring serves other needs as well (Karr 1991; Karr and Chu 1999): diagnosing the causes of degradation, evaluating restoration efforts (Yoder 1998; Yoder and Rankin 1998), and providing information on status and trends about the ''infrastructure'' of the aquatic environment.

Safe highways are maintained by tracking the condition of the infrastructure (i.e., road surface, roadbed, berms, bridges, rest areas, culverts, and drainage systems), but the state highway patrol, whose principal function is to monitor and enforce speed limits and other laws, is not charged with monitoring and maintaining the roadway infrastructure. We suggest that state water resource agencies devote equal efforts to tracking resource condition as to enforcing compliance. Today, the ''highway patrol'' function dominates at the expense of tracking and maintaining the ecological infrastructure, and little has been done to integrate the two.

Lack of attention to monitoring and assessment capacity is why state and national summary reports contain little of real value. First, only a small percentage of waters (19%) are assessed for quality, and only a limited number of assessments are based on current information (GAO 2003a). In 1994 only about 11% (630,000 out of 5.6 million km of streams and rivers) included biotic assessment (''aquatic life use attainment'') (Davis et al. 1996). By 2001 assessment rates increased to 13% (713,000 km) (U.S. EPA 2002a). At this rate, another 19 years will pass before half the nation's river miles will be assessed using biological data. Variation among the states is high in percentage of stream kilometers biologically assessed: Alabama (1.1%), Michigan (9.2%), Ohio (28.6%), and Washington (0%) for the 1995 report.

Second, even when states have ''aquatic life'' goals in their water quality standards, most measure attainment with chemical criteria, assuming incorrectly that these are reliable indicators of biological condition; they are not. Using chemical criteria, the extent of river degradation is estimated at approximately 25% of surveyed river kilometers; in contrast, 50% of river kilometers are judged impaired when biological condition is measured directly (Davis et al. 1996).

Third, selection of sampling sites without an appropriate statistical design yields misleading results. For example, in Oregon nonrandom as opposed to random approaches led to three- to-fivefold overestimates of coho salmon (*Oncorhynchus kisutch*) (Hughes et al. 2000). Fourth, even when data are available, they are so incomplete that it is difficult for EPA to accurately describe the condition of the nation's waters (GAO 2003a). Yet despite such serious flaws, these analyses are used to allocate Clean Water Act funds among the states.

In response to repeated criticisms that estimates of impaired waters are inaccurate (GAO 2000), EPA and state agencies often focus narrowly on only one dimension of the problem, determination of *general* status and trends. As a result, EPA stresses the importance of study designs that will deliver statistically valid estimates of status without the need to intensively monitor all waters (Larsen et al. 2001). But important information required for effective, place-specific TMDLs cannot be obtained with such designs, especially when state management duties and responsibilities demand information produced at different scales and intensities. The TMDL process frequently demands information at watershed- and site-specific scales, particularly in areas with multiple, overlapping stressors. States that do not monitor watersheds at thoughtfully defined geographic scales risk missing the opportunity to better understand the effects of specific human actions. In addition, site- and waterbody-specific information is essential to proper use attainability analyses, an increasingly important component of the TMDL process. For monitoring to fulfill essential support functions with a focus on end outcomes (see Figs. 1 and 2), adaptability, and diversity in spatial monitoring design are crucial.

The lack of comprehensive assessment information suggests the need to alter the balance of resources between administrative, planning, and permitting activities and monitoring. A minimum of 15–20% of state water program resources dedicated to monitoring and assessment seems appropriate. Without consistent and dedicated resources, agencies cannot accurately document the effect of their actions. Comprehensive assessments would yield two significant results. First, they would shift the focus to end outcomes (i.e., effectiveness monitoring). Second, they would build the professional and intellectual capacity to improve state programs that ultimately depend on monitoring, including WQS, TMDLs, and permitting. Without quality information, the TMDL process will be ill informed at the same time as it squanders limited financial resources, allows continuing degradation of water resources, and invites stakeholder dissatisfaction with program results.

The simple answer is to provide objective and unbiased information on a routine basis and to make it available when it is needed to improve decisions about impairment, its severity and extent, and associated causes and sources. Accomplished on a watershed scale, monitoring fulfills the information needs of multiple programs and issues. From consideration of narrow issues like point source pollution and physical habitat degradation to the multidimensional effects of urbanization and agriculture, applied research is essential to improve our understanding of the causal pathways from stressors to water resource condition. Applied programs enhance our ability to recognize patterns in environmental data as well as relationships to management programs (Morley and Karr 2002; Yoder and DeShon 2003).

Scholarly work over the past two decades by state and federal agencies, academia, and NGOs holds a number of important water resource lessons. A few examples include work in Ohio (Yoder and Rankin 1995b; Miltner and Rankin 1998; Yoder et al. 2000; Yoder and DeShon 2003), Wisconsin (Lyons 1992; Wang et al. 2000), Canada (Steedman 1988), the Pacific Northwest (Hughes et al. 1998; Morley and Karr 2002; Mebane et al. 2003), and Japan (Karr and Rossano 2001). So does public health science which successfully links scientific advances with clinical experience and observation (Karr and Rossano 2001). It is long past time to use all these lessons to improve the TMDL process.

Refine Designated Uses

Ideally, designated uses and criteria serve as rigorous guides to water resource decision making; in reality, they often do not. Nonspecific terms such as ''protection of warmwater aquatic life'' or ''warmwater fisheries'' lack the rigor necessary to reflect water-body characteristics that go beyond nationally prescribed water quality criteria. General uses and criteria simply cannot meet the demands of the TMDL process and are a principal reason for skepticism about the reliability of listings of impaired waters. Some states have already developed refined designated use frameworks: Ohio (Yoder and Rankin 1995a), Maine (Courtemanch 1995), and Vermont [Vermont Department of Environmental Conservation (DEC) 2001]. Ohio EPA developed its approach to multiple aquatic life uses, a gradient of biological condition, and numerical biological indexes that vary with ecoregion and stream or river size in the 1980s (see Fig. 5). Maine and Vermont employ a conceptually similar biological condition gradient with technically different biological assessment mechanisms from Ohio's. More states are developing similar approaches. But 13 years after the adoption of program guidance by U.S. EPA (1990), the designated use program remains voluntary, and most states simply choose not to pursue this improved methodology.

Using biological criteria and regional reference networks to improve chemical and physical criteria should also resolve many inaccurate 303(d) listings. When properly implemented, aquatic life designated uses require restrictive management strategies to halt damage from all forms of pollution. Because aquatic life uses apply to all jurisdictional waters, and the criteria apply under the most critical conditions, they are the most common ''drivers'' of water quality management.

Strengthen State and Federal Programs

Efforts to make the TMDL process credible have ranged from fostering collaboration and the collection of more samples to development of more-detailed quantitative models and recruitment of unskilled volunteers. But these efforts have not altered two fundamental problems with the TMDL program: lack of professional capacity in state programs and lack of administrative commitment in federal programs to advance seamless and integrative water resource management. As the custodians of monitoring, assessment, and WQS programs, states must set as a top priority to strengthen personnel and logistics to achieve the right mix of expertise, experience, indicators, and criteria. The nuts and bolts underlying this process are comprehensive monitoring and assessment (Karr 1991; Yoder 1998; Karr and Chu 1999), sufficiently detailed WQS, and a mix of programs and institutional priorities to accomplish water quality goals.

The second need is management support. The GAO (2003a) calls for EPA to ''ensure that recent steps to improve environmental information receive sustained top management support.'' The GAO also calls for management to embrace a focus on the measurement of environmental outcomes rather than administrative actions. Unfortunately, EPA's response thus far emphasizes reducing administrative backlogs, improving data management, and increasing assessed waters via extrapolation. It is still not an EPA priority to incorporate 3 decades of applied research and management experience that integrates biological assessment into WQS and monitoring.

When monitoring programs are driven by public perceptions, they often emphasize easily understood, ''cheaper and faster,'' or narrowly defined concepts such as water clarity, presence of pathogens, lethality, cancer-inducing toxicity, or other human health threats. While some of these are legitimate issues, they should not be the exclusive drivers of water programs, especially in view of recent advances in understanding of interactions between human activities and water resources. Those advances have unequivocally shown the need for systematic monitoring and assessment, which leads to broader, more accurate, and more representative (i.e., better) indicators, criteria, and policies. High-quality information on water resource condition leads to decisions driven less by political distortion of consequences and more by knowledge of real environmental results. Agency leadership is crucial as we strive to protect the sustainability of both the parts and processes of water bodies.

Another key challenge is to overcome the institutional tendency to ''equate'' biological, chemical, and physical criteria and indicators despite abundant evidence documenting otherwise (Karr 1991; Yoder and Rankin 1998; Karr and Chu 1999; NRC 2001). Failure seems inevitable when programs ignore these ''unequals'' and substitute, for example, chemical surrogates without considering the complementary biological context. These problems can be limited by adhering to the stressor, exposure, and response roles of indicators (see Figs. 1 and 2).

Society can no longer afford the simplicity of TMDLs framed by single pollutants and univariate condition assessment. Modern water resource assessment and management requires states to employ a more refined approach. From methods and indicators to sampling design and analysis, EPA and state agencies must lead in water management programs if the TMDL program is to become effective. A major step in that direction will come with adoption of recent scientific advances to guide the next generation of TMDLs.

Summary

A truly effective TMDL process would make protecting and restoring the integrity of water bodies its organizing principle. It would provide a seamless approach to water resource management founded on a more comprehensive conceptual model of the determinants of water resource condition. This conceptual model should explicitly do the following: use a more realistic and tractable classification of water bodies; consider the condition of water bodies in terms of a continuum of biological condition; modernize how to measure and characterize impairment and how to list waters under CWA section 303(d); incorporate the full range of stressors (all forms of pollution including pollutants); understand the cumulative effects (including interactions) of all five features upon which biological integrity depends; integrate the interaction of natural environment gradients and varying complexes of human influence; understand the chain of connections from human activities as stressors, to exposure and its effects (biological responses), including short-term effects, midterm consequences, and the more insidious and typically long delayed aftermath. It would also improve actions to restore impaired waters, evaluate the effectiveness of those actions, and streamline how 303(d)-listed water bodies are removed from those lists. Finally, cost-effective TMDLs require collaboration among state, federal, and local institutions and civil society—plus real leadership from federal and state agency managers.

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