Section

A

An Overview of Water Quality in Ohio

Clean water is important to Ohio's economy and standard of living.

Ohio is an economically important and diverse state with strong agriculture, manufacturing, and service industries. Ohio is also a water-rich state bounded by Lake Erie on the north and the Ohio River on the south, with more than 25,000 miles of named and designated streams and rivers within its borders. The suitability of these waters to support society's needs for water supplies and recreation is critical to sustaining Ohio's economy and the standard of living of Ohio citizens. Surface waters—rivers, streams, lakes—provide the majority of water used for public drinking water; for recreation such as swimming, boating, and fishing; and for industrial uses including manufacturing, power generation, irrigation, and mining.

Ohio EPA monitors water quality in Ohio and reports its findings.

Monitoring the quality of Ohio's valuable water resources is an important function of the Ohio Environmental Protection Agency. Since the early 1970s, Ohio EPA has measured the quality of Ohio's water resources and worked with industries, local governments, and citizens to restore the quality of substandard waters. The Agency reports its findings through meetings and reports. This particular report is required by the federal Clean Water Act to fulfill two purposes:

- to provide a summary of the status of the State's surface waters
- to develop a list of waters that do not meet established goals—the "impaired waters."

Under the Clean Water Act, once impaired waters are identified the state must take action to improve them. Typically, the actions include developing restoration plans [total maximum daily loads (TMDLs)], water quality based permits, and nonpoint source pollution control measures. As such, this report is an important document that provides information and direction to much of the State's work in water quality planning, monitoring, financial and technical assistance, permitting, and nonpoint source programs. The report is updated every two years.

For nearly 40 years, Ohio EPA has developed innovative monitoring methods that directly measure progress toward the goals of the Clean Water Act. Generally recognized as a leader in water quality monitoring, Ohio uses the fish and aquatic insects that live in streams to assess the health of Ohio's flowing waters. Aquatic animals are generally the most sensitive indicators of pollution because they inhabit the water all of the time. A healthy stream community is also associated with high quality recreational opportunities (e.g., fishing and boating). Stream assessments are based on the experience gained through the collection of over 26,000 fish population samples and nearly 12,500 aquatic insect community samples.


Ohio EPA biologists collect fish samples

In addition to biological data, Ohio EPA collects information on the chemical quality of the water (nearly 200,000 water chemistry samples), sediment, and wastewater discharges; data on the contaminants in fish flesh; and physical habitat information about streams. Taken together, this information identifies the factors that limit the health of aquatic life and that constitute threats to human health.

Results show water quality is impaired but continues to improve.

Ohio EPA developed methods to determine how well Ohio's waters support four specific uses of water: human health impacts related to fish tissue contamination, recreation, human health impacts related to drinking water, and aquatic life (fish and aquatic insects). Available data were compared with established water quality goals, and the results of the comparison indicate which waters are meeting goals and which are not. The results for each use are discussed in the next few pages.

To assess the **human health impacts related to fish tissue contamination**, Ohio EPA uses the same data that are used to generate Ohio's sport fish consumption advisory. Although the data are the same, the analyses are different. Ohio EPA urges Ohio's anglers to consult the sport fish consumption advisory regarding which and how much fish to eat.

For analysis in this report, approximately half of Ohio's watershed assessment units (WAUs) and two-thirds of publicly owned lakes have some fish tissue data available. Of those, about 8% of the WAUs and one-third of the lakes do not have enough data to determine the impairment status. About one-third of the monitored WAUs are "unimpaired" for the contaminants, while almost two-thirds of the WAUs are "impaired." For lakes, almost one in ten is impaired while more than half are not impaired by the six fish tissue contaminants.

The most common contaminant is polychlorinated biphenyls (PCBs), followed by mercury. A few waters contain fish whose flesh is contaminated by dichlorodiphenyltrichloroethane (DDT), mirex, or hexachlorobenzene; data show no streams or lakes with fish contaminated by lead. PCB contamination is widespread usually because of historical sources. Areas with attributable contamination and areas of special concern are being addressed through programs such as the Great Lakes Legacy Act, Superfund or the Resource Conservation and Recovery Act.

Are fish safe to eat?

While most Ohio sport fish are safe to eat, low levels of chemicals like polychlorinated biphenyls (PCBs) and mercury have been found in some fish from certain waters.

To help protect the health of Ohioans, the Ohio EPA in conjunction with the Ohio Department of Health offers an advisory for how often these fish can be safely eaten. An advisory is advice, and should not be viewed as law or regulation. It is intended to help anglers and their families make educated choices about where to fish, what types of fish to eat, how to determine the amount and frequency of fish consumed, and how to prepare fish for cooking.

By following these advisories, citizens can gain the health benefits of eating fish while reducing their exposure to unwanted contaminants.

Mercury contamination is ubiquitous because of aerial deposition from local, regional and global sources. Thus, solving the problem of mercury contamination requires solutions on a broader scale than at a watershed level. Ohio is targeting mercury from consumer products such as switches and thermometers through legislation banning the sale of such products. Ultimately, increases in renewable energy sources and clean coal technology usage will lessen Ohio's mercury burden.

Fish populations contaminated by hexachlorobenzene, DDT and mirex are already in the process of being restored through various initiatives in state and federal waste remediation programs.

The **recreation** analysis focuses on the amount of bacteria in the water. For Lake Erie public beaches, the frequency of swimming advisories varies widely, ranging from 0 percent at Kelleys Island State Park beach to over 40% at Edson, Euclid, Lakeshore, Lakeview, and Villa Angela beaches. Generally, beaches located near population centers tend to have the most problems.

Beaches on the Lake Erie islands are nearly always suitable for swimming. Several beaches stand out as consistently good performers over the past several recreation seasons, including Battery Park, Catawba, Cranberry Creek, East Harbor, Fichtel Creek, Hoffman, Kelleys Island, Lakeside, Old Womans Creek, South Bass Island, and Walnut Beach. These beaches infrequently exceeded the goal of fewer than 10 days per season under advisement. There were also several beaches

that performed poorly on a consistent basis, with five beaches (Edson Creek, Euclid, Lakeshore, Lakeview, and Villa Angela) under advisement for more than 40% of the past five recreation seasons.

For inland streams, bacteria levels were low in about one in ten watersheds. About three in ten watersheds had high levels of bacteria. The remaining six in ten did not have enough data for evaluation. Ohio's 23 large rivers fare somewhat better, with about 20 percent having relatively low bacteria levels and 20 percent showing higher levels of bacteria. About 60 percent did not have enough data collected in the past five years to evaluate. High bacteria levels are often observed during periods of higher stream flows associated with heavy rains.

Is it safe to swim or wade?

For the most part, water in Ohio is safe for swimming or wading. Water activities are more dangerous after heavy rains due to the obvious physical dangers of being swept into the faster flows, but also because chemicals and bacteria wash into the streams along with the water that runs over the land. In some communities, sewage systems cannot handle the extra volume of water and release untreated sewage during and after heavy rains.

There are some areas where the waters and/or sediments have high levels of contaminants, including polychlorinated biphenyls (PCBs) and polyaromatic hydrocarbons (PAHs), so swimming or wading in these areas is not recommended.

Although not sampled as frequently as streams or Lake Erie beaches, bacteria levels at most inland lake beaches do not frequently exceed the threshold, resulting in fewer postings compared to some of the beaches along Lake Erie.

Is water safe to drink?

Yes. Public water systems around the state and Ohio EPA work hard to ensure that the water provided meets safe drinking water standards and to make important information available about the sources and quality of the water you drink. However, drinking water advisories do occur from time to time due to treatment plant malfunctions, water line breaks, and the rare case when source water contaminant levels exceed the plant's capacity to remove them. It is important to remember that only a relatively small number of water systems have situations that warrant advisories. In 2010, 99% of all public water systems met all chemical standards. In order to get information about your local drinking water you can read the Consumer Confidence Report (CCR) provided annually by your community water system.

In this report several waters are identified as impaired due to elevated nitrate or pesticides. Water systems in these areas and others with source water contaminants will issue public notice advisories or use additional treatment and water management strategies to assure that safe water is delivered to their customers.

Human health impacts related to drinking water focus on nitrate and pesticides, and for the first time in 2014, cyanotoxin (due to certain algae). There are a total of 119 public water systems using surface water (excluding Ohio River intakes). Sufficient data were available to evaluate about one-third of the drinking water source waters for nitrate.

The only impaired areas were the Maumee River (the systems for the communities of Defiance, Napoleon, McClure and Bowling Green and the Campbell Soup system) and a portion of the Sandusky River (Fremont). Some areas were identified for a watch list; most were located in the northwestern and central parts of the state. It is difficult and expensive to remove nitrate from drinking water; some systems are conducting nitrate removal pilot studies, but no Ohio surface water systems currently use treatment specific for nitrate removal. Ohio public water systems rely on blending the surface water with other sources such as ground water, selective pumping from the stream to avoid high nitrate levels by using off-stream storage in upground reservoirs, or issue public notice advisories warning sensitive populations to avoid drinking the water while nitrate levels are high.

Pesticides could be evaluated for about 14 percent of the drinking water source waters. Five of 18 areas were identified as impaired, all in southwestern Ohio: one in Brown County (Mt. Orab), one in Miami County (Piqua), and the three sources used by the Village of Blanchester in Warren and Clinton counties. Thirteen areas were identified for a watch list because of elevated atrazine. These areas mostly coincide with the predominantly agricultural lands of western and northwestern Ohio.

In recent years, algae (cyanotoxin) data have been collected in response to harmful algal blooms. Based on this data, impairments were identified in source waters of public drinking water systems for Celina, Clermont County, Akron, Lima, Oregon, Carroll Township, Ottawa, Toledo, and Marblehead. Over half of the water systems with impaired source waters draw water from the western basin of Lake Erie.

The bulk of the new data evaluated for the **aquatic life use** is in areas Ohio EPA sampled during 2011 and 2012. Watersheds intensively monitored during 2011 and 2012 included Tenmile Creek, Deer Creek, the upper Little Miami River, the Ashtabula River, the lower Scioto River, the Black River, Stillwater Creek, Mill Creek (in the Scioto River basin), the East Fork Little Miami River, and the large river mainstems of the Maumee River, the Auglaize River and the Tiffin River. Detailed watershed survey reports for many of these watersheds are or will be available at http://epa.ohio.gov/dsw/document_index/psdindx.aspx.

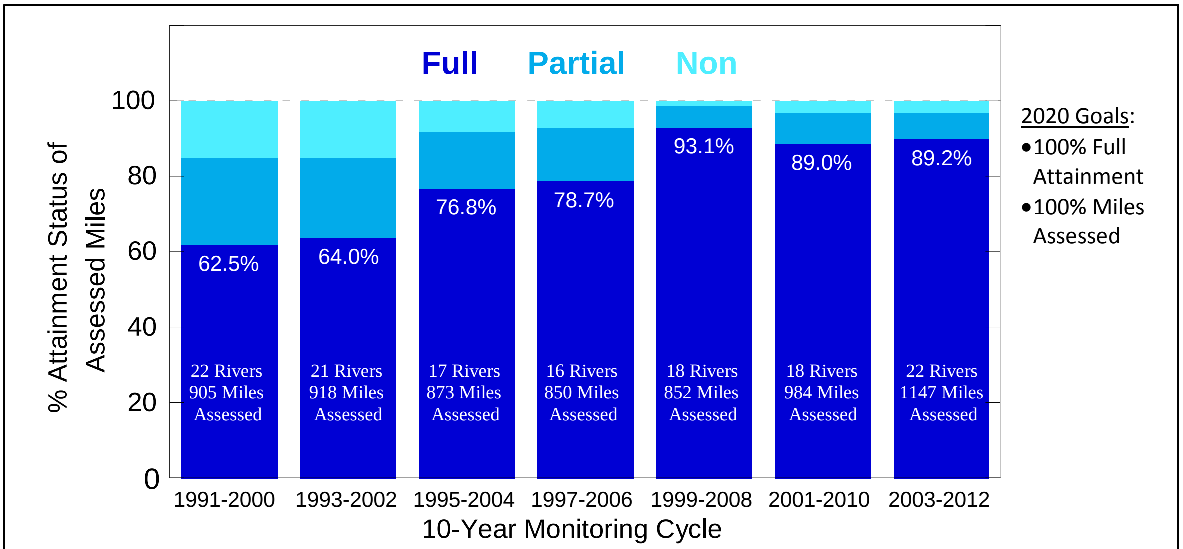
Large rivers are making progress towards the "100% attainment by 2020" aquatic life goal.

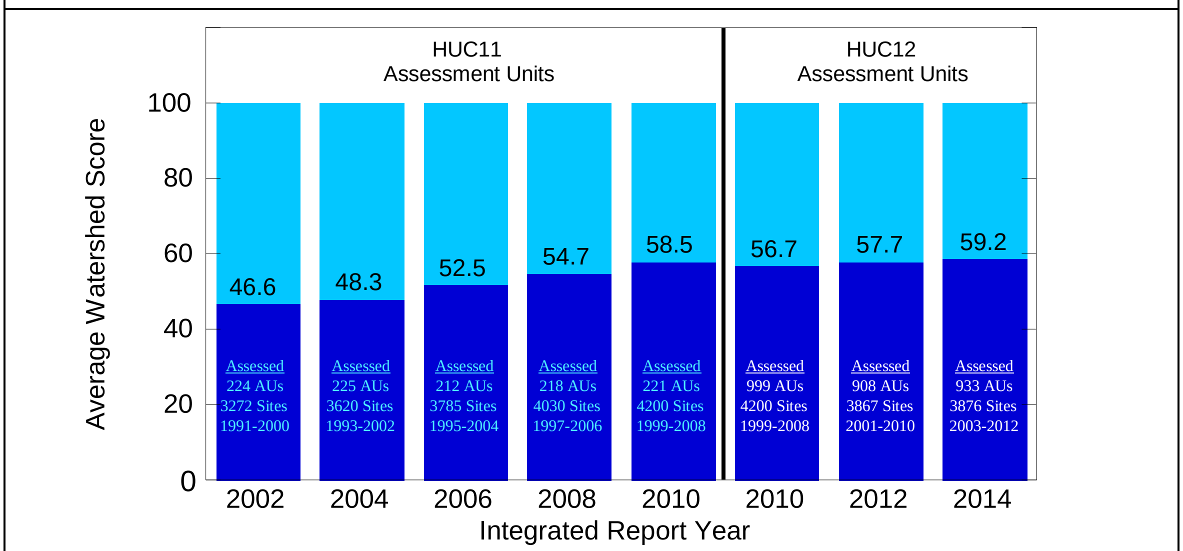
Ohio's large rivers (the 23 rivers that drain more than 500 square miles) continue to show improvement as tracked over the last 20 years. The "100% full attainment by 2020" aquatic life goal statistic remains steady at 89.2 percent full attainment. The table below shows the status of the four large rivers recently sampled, particularly the improvement in the Maumee and Tiffin Rivers since the mid to late 1990s.

Stream	Year Studied	% of Stream Monitored	% of Aquatic Life Standard			
			Meeting	Partially Meeting	Not Meeting	Not Known
Maumee River	1997	89	25	28	47	11
	2012	100	81	5	14	0
Tiffin River	1992	100	0	100	0	0
	2012	100	100	0	0	0
Auglaize River	2000	100	100	0	0	0
	2012	100	100	0	0	0
Scioto River (Big Darby Creek to Ohio River)	1997	100	92	8	0	0
	2011	100	100	0	0	0

Taken collectively since the 1980s, the quality of aquatic life in all of Ohio's large rivers has shown a remarkable improvement. Then, only 21 percent of the large rivers met water quality standards, increasing to 62 percent in the 1990s, to 89 percent today. Areas not meeting the standards have decreased from 79 percent in the 1980s to 38 percent in the 1990s to 11 percent today. Across Ohio, investment in the treatment of municipal and industrial wastewater and improvement in agriculture conservation practices are credited with the turnaround. The substantial aquatic life improvements observed in these rivers over the last 25 years directly correlate to implementation of agricultural best management practices and upgraded wastewater treatment plants. Being able to track these water quality trends attests to the value of consistent monitoring over time.

For Ohio's 1,538 12-digit hydrologic watershed units, the score calculated from measurements at individual sites also continued its steady increase, although with an average score considerably lower than the large river full attainment statistic. Watershed scores are roughly equivalent to the percent of sites within the watershed unit that are meeting biological expectations and the designated aquatic life use, but some additional weighting is given to results from larger stream sites in the unit. Based on monitoring through 2012, the average watershed score is now 59.2 (of watersheds with data), up from 57.7 in 2012. Of the 933 watershed units assessed for this report with current data, 418 (45 percent) scored 80 or above and 341 (37 percent) scored perfect 100s. The following charts show the progress in attainment status of aquatic life statistics in recent years for both large rivers (upper) and watersheds (lower).



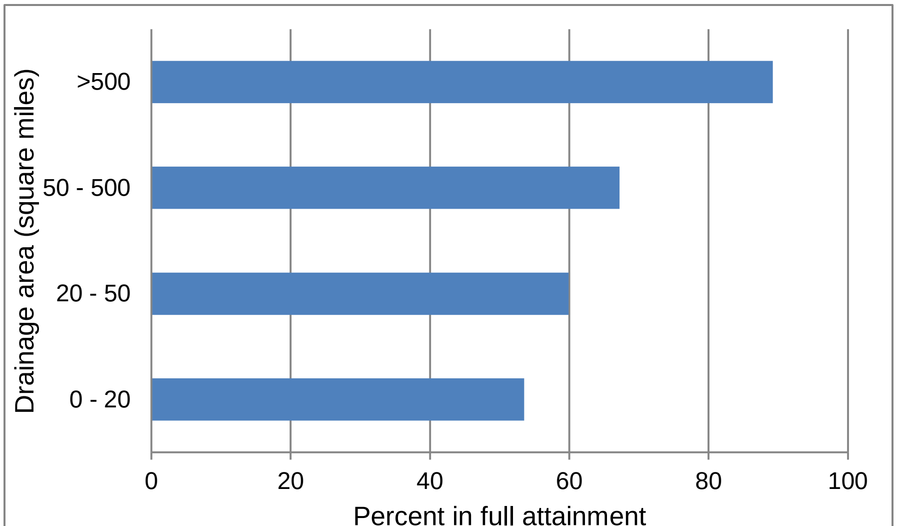


The collection of more biological data along the shore of Lake Erie as a result of the Great lakes Restoration Initiative allows a more current analysis of shoreline conditions. The aquatic life use of the

Lake Erie shoreline is impaired due primarily to tributary loadings of nutrients and sediment, aggravated by the proliferation of exotic species, algal blooms, and shoreline habitat modifications.

Most aquatic life impairment is caused by land disturbances related to agriculture activities and urban development.

Taking a closer look at the attainment status of individual sites grouped by the amount of land area drained by the stream at that point reveals that unhealthy fish and aquatic insect populations are more common on smaller streams (see chart below). In other words, the larger the drainage area (and usually the larger the stream), the more likely the stream is to be healthy. This phenomenon correlates well with the most widespread causes associated with the aquatic life impairment in these watersheds.



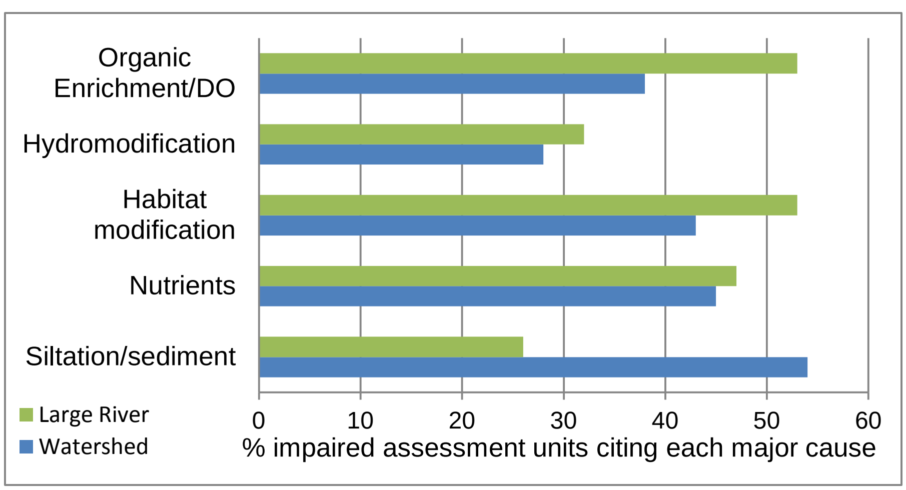
For 0 to 500, based on number of sites in attainment; for > 500, based on number of miles.

The top five aquatic life impairment causes for the period 2003 through 2012 are:

- siltation/sedimentation
- nutrients
- habitat modification
- hydromodification
- organic enrichment / dissolved oxygen (DO).

For watersheds, most impairment is related to modification of the landscape. These types of impairments have the most impact on smaller streams. Most of the impaired watershed units with current data had at least one of these causes contributing to impairment and many had two or more of the top five causes listed.

Of note is the prevalence of watersheds and large rivers that are impaired by the generic organic enrichment/DO cause category; 38 percent of impaired watersheds show "sewage" related impairments such as high biochemical oxygen demand, elevated ammonia concentrations, and/or in-stream sewage solids deposition. Ten of 19 impaired large rivers also note sewage-related causes. This suggests that adequate treatment and disposal of human and animal wastes via wastewater treatment plants, home sewage



treatment systems, and land applications of septage and animal manure continue to be critical water quality issues in many Ohio watersheds.

The major causes and sources of water quality problems are described below.

Organic enrichment is the addition of carbon-based materials from living organisms beyond natural rates and amounts. Natural decomposition of these materials can deplete oxygen supplies in surface waters. Dissolved oxygen is vital to fish and other aquatic life and for the prevention of odors associated with the decomposition process.

Siltation/sedimentation describes the deposition of fine soil particles on the bottom of stream and river channels. Deposition typically follows high-flow events that erode and pick up soil particles from the land. Soil particles also transport other pollutants. As the flow decreases, the soil particles fall to the stream bottom. This reduces the diversity of stream habitat available to aquatic organisms.

Nutrient enrichment describes the excess contribution of materials such as nitrogen and phosphorus used for plant growth. Excess nutrients are not toxic to aquatic life, but can have an indirect effect because algae flourish where excess nutrients exist. The algae die and their decay uses up the dissolved oxygen that other organisms need to live. The aquatic community is stressed on both a daily basis and over the long term.

Habitat modification is the straightening, widening, or deepening of a stream's natural channel. Habitat modification can also include the degrading or complete removal of vegetation from stream banks; such vegetation is essential to a healthy stream. These activities can effectively transform a stream from a functioning ecosystem to a simple drainage conveyance. Some aquatic life will not be protected from predators and stressful flows and temperatures. The stream also often loses its ability to naturally process water pollutants.

Hydromodification, or flow alteration, describes any disruption to the natural hydrology of a stream system. Flow alteration includes stream impoundment, increased peak flows associated with the urbanization of watersheds, and water-table regulation through sub-surface drainage. Such changes can cause extended periods without stream flow, more extreme or frequent floods, and loss of fast current habitat in dam pool areas.

Contamination by pathogens occurs when human or animal waste reaches the stream. Pathogenic organisms include bacteria, viruses, and protozoa. Contamination by pathogens is a human health issue, as skin contact or accidental ingestion can lead to various conditions such as skin irritation, gastroenteritis, or other more serious illnesses.

Excessive nutrients lead to excessive algae growth.

The same nutrients that cause impairment of the aquatic life beneficial use also are a major contributing factor to the recent extensive harmful algal blooms (HABs) that have been observed in Lake Erie, the Ohio River, and many inland Ohio water bodies. Grand Lake St. Marys in western Ohio has been particularly affected. HABs, a visually identified concentration of cyanobacteria, can occur almost anywhere there is water: lakes, ponds, storm water retention basins, rivers, streams, or reservoirs. Many HAB-forming organisms are native to Ohio but only cause problems when environmental conditions favor them.

Harmful algal blooms can cause taste and odor problems in drinking waters, pollute beaches with scums, reduce oxygen levels for fish and other animals, cause processing problems for public water supplies, and may generate toxic chemicals. Knowing what triggers HABs is key to reducing their occurrence and impacts. HABs may be minimized, and some completely avoided, by reducing the nutrients and pollutants added to the water.

The Ohio EPA, the Ohio Department of Health (ODH) and the Ohio Department of Natural Resources (ODNR) developed a strategy to protect people from toxins produced by cyanobacteria that may be in recreational waters at concentrations that can affect human health. The report outlines thresholds for identified algal toxins, establishes monitoring protocols and identifies the process for posting and removing recreation use advisories. A web site was established to provide background information about HABs, tips for staying safe when visiting public lakes, links to sampling information and current advisories and contact information for reporting suspected HABs.

Understanding how various land uses impact water quality can lead to more effective prevention and restoration.

Ohio has embraced a wide variety of economic enterprises over the past 150 years, so it is not surprising that there is a large variety of causes and sources of impairment.

Row crop cultivation is a common land use in Ohio. Frequently, cultivated cropland involves tile drainage, and a challenge is to carry out actions that improve water quality while maintaining adequate drainage for profitable agriculture. The land application of manure, especially during winter months, is often a large source of both bacteria and nutrients entering streams and subsurface drainage tiles. Many cropland practices involve the channelization of streams, which creates deeply incised and straight ditches or streams. This disconnects waterways from floodplains, which has damaging impacts on the quality of the system. The regularity of the stream channel and lack of in-stream cover reduces biological diversity.

Land development is the conversion of natural areas or agriculture to residential, industrial, or commercial uses. Numerous scientific studies show that increasing impervious cover—hard surfaces such as roads, parking lots, rooftops, and lawns—harms water quality. More water runs off the hard surfaces and more quickly. The rate of erosion increases and streams become unstable. The resulting channel is less able to assimilate nutrients and other pollution. Higher runoff volume increases the amount of pollutants (e.g., nutrients, metals, sediment, salts, pesticides). Another problem is that stream temperatures can be raised when water runs over hot pavement and rooftops or sits in detention basins. When this heated water enters a stream, the higher temperatures reduce dissolved oxygen concentrations that aquatic life need to survive. With proper planning of development, many of these problems can be mitigated or avoided entirely.

Agricultural livestock operations can vary widely in how they are managed. Pasture land and animal feeding operations can be sources of nutrients and pathogens. Frequently livestock are permitted direct access to streams. Direct access not only allows direct input of nutrients and pathogens, but also erodes the stream bank, causing excess sediments to enter the stream and habitat degradation. The most critical aspect of minimizing water quality impacts from any size animal feeding operation is the proper management of manure in terms of application and storage.

Industrial and municipal point sources include wastewater treatment plants and factories. Wastewater treatment plants can contribute to bacteria, nutrient enrichment, siltation, and flow alteration problems. Industrial point sources, such as factories, sometimes discharge water that is excessively warm or cold, changing the temperature of the stream. Point sources may contain other pollutants such as chemicals, metals and solids.

Acid mine drainage impacts streams with high levels of acidity (low pH), high metal concentrations, elevated sulfate levels, and/or excessive dissolved and suspended solids and/or siltation. Acid mine drainage often has toxic effects on stream organisms and degrades habitat quality when deposited metals form a crust on the stream bed and susceptible soils erode from areas disturbed from mining. Ultimately it reduces biological diversity, eliminates sensitive aquatic life, and lowers ecosystem productivity.

Solving Ohio's water quality problems will require collaboration and creativity.

Most of Ohio's water quality problems will not be solved by issuing a permit or building a new wastewater treatment system to treat point sources of pollution. Improving Ohio's surface water quality will require effectively managing land use changes to ensure that polluted runoff is either captured and treated or allowed to infiltrate through the soil before running off into a stream. Restoring and protecting natural stream functions so that pollutants may be more effectively assimilated by streams is also critical. These actions will require various programs and people working collaboratively on local water quality issues and concerns. Local educational efforts and enhanced water quality monitoring will also play important roles if we are to see significant water quality improvements throughout Ohio.

Many areas of the state are benefitting by the participation of individuals and organizations in local watershed organizations. Some of these organizations have been active for quite some time and are successfully influencing local land use decision making and implementing projects designed to improve water quality in their watershed. Since 2000, Ohio EPA has worked in conjunction with the Ohio Department of Natural Resources to provide section 319(h) grant funding assistance to hire local watershed coordinators to help facilitate the development of watershed action plans. In recent years, the emphasis has shifted from developing plans to implementing water quality improvement projects such as stream restoration, dam removals, agricultural best management practices and others. Ohio EPA is measuring improvements resulting from these projects; however, there remain challenges associated with changing land use decisions and consumer and producer attitudes.

The report provides more detail, including Ohio's Section 303(d) list of impaired waters, as required by the Clean Water Act.

This overview is intended to provide a snapshot of water quality conditions, progress and challenges in Ohio; it is only the first section of the much larger and more detailed 2014 Integrated Report.

The opening sections of the report describe the universe of water quality in Ohio—the size and scope of Ohio's water resources, programs that are used to evaluate and improve water quality and funding sources for water quality improvement.

The middle sections are more technical and explain the beneficial uses assigned to Ohio's waters, the assessment methodologies used for the analyses of those uses, the data used to determine whether those uses are being supported, and the conclusions drawn about water quality conditions in each assessment unit.

The closing sections describe how waters found to be impaired will be scheduled for further study. A collection of maps that illustrate current conditions and future plans follow the text. The report concludes with summary tables of various types. The 303(d) list is contained in Section L4. Summaries of the condition of each assessment unit are available at http://wwwapp.epa.ohio.gov/gis/mapportal/IR2014.html.

For more information, please consult these web sites:

Many water quality reports on specific watersheds are mentioned in this overview. Find these reports at http://www.epa.ohio.gov/dsw/document_index/psdindx.aspx

Watershed restoration reports (TMDLs) … http://www.epa.ohio.gov/dsw/tmdl/index.aspx

Fish consumption advisory … http://www.epa.ohio.gov/dsw/fishadvisory/index.aspx

Harmful algal blooms … www.ohioalgaeinfo.com

Integrated Report … http://www.epa.ohio.gov/dsw/tmdl/OhioIntegratedReport.aspx

Ohio EPA Division of Surface Water … http://www.epa.ohio.gov/dsw/SurfaceWater.aspx

Ohio EPA Division of Drinking and Ground Waters … http://www.epa.ohio.gov/ddagw/DrinkingandGroundWaters.aspx

Ohio EPA district office contact info … http://www.epa.ohio.gov/directions.aspx

List of Ohio watershed groups … http://ohiowatersheds.osu.edu/groups/

Ohio Department of Natural Resources, Division of Soil and Water Resources … http://www.dnr.state.oh.us/tabid/21817/Default.aspx

U.S. Environmental Protection Agency water program … http://water.epa.gov/

Exhibit 52 is not publicly posted on the MPCA web page due to copyright protection laws. However, the following link is provided for interested parties to access the document in accordance with the respective copyright restrictions. The document may also be available through your local library.

Karr J. R. (1999) Defining and measuring river health. *Freshwater Biology* 41: 211-234.

http://onlinelibrary.wiley.com/doi/10.1046/j.1365-2427.1999.00427.x/abstract

Exhibit 53 is not publicly posted on the MPCA web page due to copyright protection laws. However, the following link is provided for interested parties to access the document in accordance with the respective copyright restrictions. The document may also be available through your local library.

Larsen D. P., J. M. Omernik, R. M. Hughes, et al. (1986) Correspondence between spatial patterns in fish assemblages in Ohio streams and aquatic ecoregions. *Environmental Management* 10: 815-828.

http://link.springer.com/article/10.1007/BF01867734

Exhibit 54 is not publicly posted on the MPCA web page due to copyright protection laws. However, the following link is provided for interested parties to access the document in accordance with the respective copyright restrictions. The document may also be available through your local library.

Herlihy A. T., S. G. Paulsen, J. V. Sickle, J. L. Stoddard, C. P. Hawkins & L. L. Yuan. (2008) Striving for consistency in a national assessment: the challenges of applying a reference condition approach at a continental scale. *Journal of the North American Benthological Society* 27: 860-877.

https://cfpub.epa.gov/si/si_public_record_report.cfm?dirEntryId=203564

Exhibit 55 is not publicly posted on the MPCA web page due to copyright protection laws. However, the following bibliographic citation is provided so that interested parties may acquire a copy of the document in accordance with the respective copyright restrictions. The document may also be available through your local library.

Hughes R. M. (1995) Defining acceptable biological status by comparing with reference conditions. In: *Biological assessment and criteria: Tools for water resource planning and decision making* (eds W. S. Davis & T. P. Simon) pp. 31-47. Lewis, Boca Raton, FL.

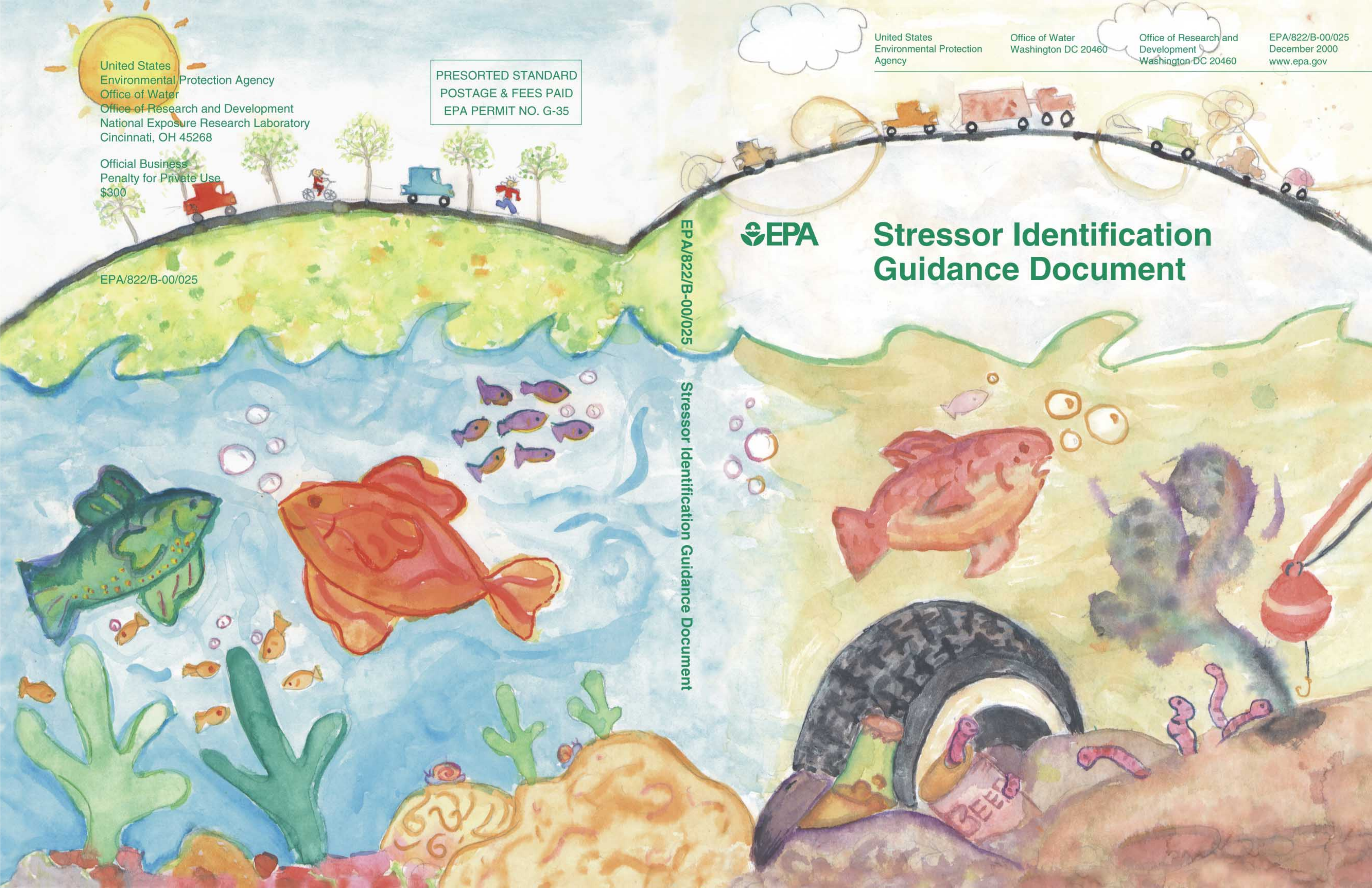
Exhibit 56 is not publicly posted on the MPCA web page due to copyright protection laws. However, the following link is provided for interested parties to access the document in accordance with the respective copyright restrictions. The document may also be available through your local library.

Davies S. P. & S. K. Jackson. (2006) The biological condition gradient: a descriptive model for interpreting change in aquatic ecosystems. *Ecological Applications* 16: 1251-1266.

http://onlinelibrary.wiley.com/doi/10.1890/1051-0761(2006)016[1251:TBCGAD]2.0.CO;2/abstract?systemMessage=Wiley+Online+Library+will+be+unavailable+on+Saturday+17th+December+2016+at+09%3A00+GMT%2F+04%3A00+EST%2F+17%3A00+SGT+for+4hrs+due+to+essential+maintenance.Apologies+for+the+inconvenience

Exhibit 57 is not publicly posted on the MPCA web page due to copyright protection laws. However, the following bibliographic citation is provided so that interested parties may acquire a copy of the document in accordance with the respective copyright restrictions. The document may also be available through your local library.

Yoder C. & J. DeShon (2003) Using biological response signatures within a framework of multiple indicators to assess and diagnose causes and sources of impairments to aquatic assemblages in selected Ohio rivers and streams. In: Biological Response Signatures: Indicator Patterns Using Aquatic Communities (ed T. P. Simon) pp. 23-82. CRC, Boca Raton, FL.



United States
Environmental Protection Agency
Office of Water
Office of Research and Development
National Exposure Research Laboratory
Cincinnati, OH 45268

PRESORTED STANDARD
POSTAGE & FEES PAID
EPA PERMIT NO. G-35

Official Business
Penalty for Private Use
$300

EPA/822/B-00/025

EPA/822/B-00/025

Stressor Identification Guidance Document

United States
Environmental Protection
Agency

Office of Water
Washington DC 20460

Office of Research and
Development
Washington DC 20460

EPA/822/B-00/025
December 2000
www.epa.gov

EPA

Stressor Identification Guidance Document

STRESSOR IDENTIFICATION GUIDANCE DOCUMENT

U.S. Environmental Protection Agency

Office of Water
Washington, DC 20460

Office of Research and Development
Washington, DC 20460

EPA-822-B-00-025

December 2000

Disclaimer

This Stressor Identification Guidance Document provides guidance to assist EPA Regions, States, and Tribes in their efforts to protect the biological integrity of the Nation's waters, one of the primary objectives of the Clean Water Act (CWA). It also provides guidance to the public and the regulated community on identifying stressors that cause biological impairment. While this document constitutes the U.S. Environmental Protection Agency's (EPA's) scientific recommendations regarding stressor identification, this document does not substitute for the CWA or EPA's regulations, nor is it a regulation itself. Thus, it cannot impose legally binding requirements on EPA, States, Tribes, or the regulated community, and may not apply to a particular situation based upon the circumstances. When appropriate, State and Tribal decisionmakers retain the discretion to adopt approaches on a case-by-case basis that differ from this guidance. EPA may change this guidance in the future.

Acknowledgments

Primary Authors:

EPA, Office of Research and Development:
Susan Cormier, Ph.D.
Susan Braen Norton, Ph.D.
Glenn Suter II, Ph.D.

EPA, Office of Science and Technology:
Donna Reed-Judkins, Ph.D.

Contributing Authors:

EPA, Office of Science and Technology:
Jennifer Mitchell
William Swietlik
Marjorie Coombs Wellman

EPA, Office of Wetlands, Oceans and Watersheds:
Thomas Danielson
Chris Faulkner
Laura Gabanski, Ph.D.
Molly Whitworth, Ph.D.

EPA, Office of Research and Development:
Edith Lin, Ph. D.
Bhagya Subramanian

EPA, Office of Enforcement and Compliance Assurance
Brad Mahanes

Other Affiliations:
David Altfater, Ohio Environmental Protection Agency
William Clements, Ph.D., Colorado State University, Fort Collins, Colorado
Susan P. Davies, Ph.D., Maine Department of Environmental Protection, Augusta, Maine
Jeroen Gerritsen, Ph.D., Tetra Tech, Owings Mills, Maryland
Martina Keefe, Tetra Tech, Owings Mills, Maryland
Sandy Page, Tetra Tech, Owings Mills, Maryland
Jeffrey Stinson, Ph.D., U.S. Air Force

Technical Editors:

EPA, Office of Research and Development, National Risk Management and Restoration Lab:
Jean Dye, Ph.D.
Scott Minamyer

Tetra Tech:
Abby Markowitz
Sandra Page
Colin Hill
Brenda Fowler

Stressor Identification and Evaluation Workgroup Members:

Co-leads:
Office of Water: Donna Reed-Judkins, Ph.D., Office of Science and Technology
Office of Research and Development: Susan Cormier, Ph.D., National Exposure Research Lab

Members:
Office of Water:
Office of Science and Technology:
Tom Gardner, Susan Jackson, Jennifer Mitchell, Keith Sappington, Treda Smith, William Swietlik, Brian Thompson, Marjorie Wellman

Office of Wetlands, Oceans, and Watersheds:
Thomas Danielson, Laura Gabanski, Chris Faulkner, Molly Whitworth, Ph.D.

Office of Research and Development:
National Center for Environmental Assessment:
Susan Norton, Ph.D., Glenn Suter II, Ph.D.

National Health and Environmental Effects Laboratory:
Naomi Detenbeck, Ph.D., Wayne Munns, Ph.D.

National Risk Management and Restoration Laboratory:
Alan Everson, Scott Minamyer

Office of Enforcement and Compliance Assurance
Brad Mahanes

EPA Regions
Toney Ott, Region 8

Other Federal Agencies:
Jeffrey Stinson, Ph.D., U.S. Air Force

States:
Susan Davies, Maine Department of Environmental Protection, Augusta, Maine
Chris O. Yoder, Ohio EPA, Columbus, Ohio

Other Supporting EPA Members:
Don Brady, Alan Hais, Margarete Heber, Mary Sullivan

Contract Support, Tetra Tech, Owings Mills, Maryland:
Michael Barbour, Ph.D., Jeroen Gerritsen, Ph.D., Martina Keefe, Sandy Page

<u>Peer Reviewers:</u>

A. Fred Holland, Ph.D., Director, Marine Resources Research Institute of South Carolina.
Kent Thornton, Ph.D., FTN Associates
Wayne Landis, Ph.D., Director, Institute of Environment Toxicology and Chemistry, Western Washington University

The authors wish to gratefully acknowledge all others, not named above, who helped to prepare this document. The sum of these efforts contributed to the success of this guidance. Special thanks also to all the EPA and State scientists who participated in the video conference in October, 1999; the Region III Mid-Atlantic Water Pollution Biology Workshop at Cacapon, West Virginia, in March 2000; and the Biological Advisory Committee meeting in Cincinnati, in May, 2000. Comments were also provided by a number of EPA scientists and regulators and by other stakeholders, including the Kansas Department of Health and Environment, Arizona Department of Environmental Quality, Denver Metro Wastewater Reclamation District, Pennsylvania Department of Environmental Protection, Proctor and Gamble, The Nature Conservancy, and the U.S. Geological Survey. Comments from the workshops and other commenters helped shape the guidance.

The cover illustration was provided by a fifth grade student at Ursula Villa Elementary School, Mount Lookout, OH. According to the illustrator, the front cover is the river when you first pick up this book, and the back cover is the river after you've followed the instructions.

Table of Contents

Table of Contents (continued)

List of Figures

Figure **Page**

List of Figures (continued)

List of Tables

Table **Page**

List of Tables (continued)

Acronym List

303(d)	The section of the Clean Water Act that requires a listing by states, territories, and authorized tribes of impaired waters, which do not meet the water quality standards that states, territories, and authorized tribes have set for them, even after point sources of pollution have installed the minimum required levels of pollution control technology.
305(b)	The section of the Clean Water Act that requires EPA to assemble and submit a report to Congress on the condition of all water bodies across the Country as determined by a biennial collection of data and other information by States and Tribes.
7Q10	Lowest average 7 consecutive days flow with average recurrence frequency of once every 10 years
BAP	Benzo[a]pyrene
BOD	Biological Oxygen Demand
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
COD	Chemical Oxygen Demand
CSOs	Combined Sewer Outfalls
CWA	Clean Water Act
DELTA	Deformities, Erosions, Lesions, Tumors, and Anomalies
DDT	Dichlorodiphenyltrichloroethane
DNR	Department of Natural Resources
DO	Dissolved Oxygen
DQA	Data Quality Assessment
DQO	Data Quality Objectives
ECBP	Eastern Cornbelt Plains
EMAP	Environmental Monitoring and Assessment Program
EPA	U.S. Environmental Protection Agency
EPT	Ephemeroptera-Plecoptera-Tricoptera
EROD	Ethoxy Resorufin[o]deethylase

FACA	Federal Advisory Committee Act
GIS	Geographic Information System
IBI	Index of Biotic Integrity
ICI	Invertebrate Community Index
IFIM	Instream Flow and Incremental Methodology
KBI	Kansas Biotic Index
KDHE	Kansas Department of Environmental Protection
MBI	Macroinvertebrate Biotic Index
MIWB	Modified Index of Well-Being
MWH	Modified Warmwater Habitat
NA	Not Applicable/Available
NAPH	Naphthalene
NE	No Evidence
ND	Not Detected
NEP	National Estuaries Program
NIH	National Institute of Health
NO_x	Nitrites
NPDES	National Pollution Discharge Elimination Act
NPS	Non-point Source
NRC	National Research Council
OEPA	Ohio Environmental Protection Agency
PAHs	Polycyclic Aromatic Hydrocarbons
PEL	Probable Effect Level
PO_4	Ortho-phosphate
POTWs	Publicly Owned Treatment Works
QHEI	Qualitative Habitat Evaluation Index

RM	River Mile
SECs	Sediment Effect Concentrations
SEP	Supplemental Environmental Protection
SI	Stressor Identification
TEL	Threshold Effect Level
TKN	Total Kjeldahl Nitrogen
TMDL	Total Maximum Daily Load
TP	Total Phosphorus
TIE	Toxicity Identification Evaluation
TRE	Toxicity Reduction Evaluation
TSS	Total Suspended Solids
USEPA	U.S. Environmental Protection Agency
WET	Whole Effluent Toxicity
WWH	Warm Water Habitat
WWTP	Waste Water Treatment Plant

Executive Summary

In this Summary:

ES.1 The Clean Water Act, Biological Integrity, and Stressor Identification

Since the inception of the Clean Water Act (CWA) in 1972, the rivers, lakes, estuaries, and wetlands of the United States have indeed become cleaner. The standard for measuring these improvements are both chemical and biological. Yet, we know that many waterbodies still fail to meet the goal of the Clean Water Act – to maintain the chemical, physical and biological integrity of the nation's waters.

Biological assessments have become increasingly important tools for managing water quality to meet the goals of the CWA. These methods, which use measurements of aquatic biological communities, are particularly important for evaluating the impacts of chemicals for which there are no water quality standards, and for non-chemical stressors such as flow alteration, siltation, and invasive species. However, although biological assessments are critical tools for detecting impairment, they do not identify the cause or causes of the impairment.

The ability to accurately identify stressors and defend the evidence supporting those findings is a critical step in developing strategies that will improve the quality of aquatic resources.

The Office of Water and Office of Research and Development of the US EPA have developed a process for identifying any type of stressor or combination of stressors that cause biological impairment. The Stressor Identification (SI) Guidance is intended to lead water resource managers through a formal and rigorous process that

- identifies stressors causing biological impairment in aquatic ecosystems, and
- provides a structure for organizing the scientific evidence supporting the conclusions.

The ability to accurately identify stressors and defend the evidence supporting those findings is a critical step in developing strategies that will improve the quality of aquatic resources.

The Stressor Identification process (SI) is prompted by biological assessment data indicating that a biological impairment has occurred. The general SI process entails critically reviewing available information, forming possible stressor scenarios that might explain the impairment, analyzing those scenarios, and producing conclusions about which stressor or stressors are causing the impairment. The SI process is iterative, usually beginning with a retrospective analysis of available data. The accuracy of the identification depends on the quality of data and other information used in the SI process. In some cases, additional data collection may be necessary to accurately identify the stressor(s). The conclusions can be translated into management actions and the effectiveness of those management actions can be monitored.

ES.2 Intended Audience

This guidance should prove useful to anyone involved in managing impaired aquatic ecosystems. The results of Stressor Identification investigations are valuable to many types of environmental managers— including land-use planners, industrial and municipal dischargers, reclamation companies, and any individuals or organizations involved in activities that directly or indirectly affect water quality or aquatic habitats.

Although the Stressor Identification process is scientifically rigorous, it is flexible enough to support various water management requirements.

The process of stressor identification draws upon a broad variety of disciplines and is most effective when the SI investigator has input from professionals in a number of environmental areas such as aquatic ecology, biology, geology, geomorphology, statistics, chemistry, environmental risk assessment, and toxicology. Sophisticated knowledge in certain fields may increase the tools available to investigators (e.g., physiological responses to certain stressors), but the SI process also can be used by investigators with very general tools (e.g., fish population estimates). Results of general measures, however, may not be as precise as when more specialized measures are used (e.g., stomach-lining histological evaluations).

ES.3 Applications of the SI Process

Although the Stressor Identification process is scientifically rigorous, it is flexible enough to support various water management requirements. Some potential applications of the SI process include the following:

- **Characterizing the Quality of the Nation's Waters:** Stressor Identification procedures can assist states in more accurately identifying the causes of biological impairment in 305(b) reporting.

- **Identifying Waterbodies and Wetlands that Exceed Water Quality Standards:** Accurate, reliable stressor identification procedures are necessary for EPA and the states/tribes to accurately identify the cause(s) of water quality standards violations for 303(d) listing and Total Maximum Daily Load (TMDL) calculations. The SI process can help achieve higher degrees of accuracy and reliability in identifying pollutants causing impacts. The SI process is not designed, however, to allocate the amount of responsibility for an impact to a particular source, especially when multiple sources of a stressor are present.

- **Regulatory and Non-Regulatory Pollution Management Programs:** Stressor identification procedures can help identify different types of stressors within a watershed that are contributing to biological impairment. Stressors can then be prioritized and controlled through a combination of voluntary and mandatory programs.

Other types of programs in which the SI process is useful include: State/Local Watershed Management Programs, National Pollutant Discharge Elimination System (NPDES) Permitting Programs, Dredge and Fill Permitting, Compliance and Enforcement Actions, Risk Assessments, Preservation and Restoration Programs, and Control Effectiveness Assessments.

If a legal challenge to the conclusions drawn is possible, or if costly remediation efforts are indicated as the means to control a stressor, it is essential to have a high level of confidence in the accuracy of the identification. However, because requirements for confidence levels and stressor precision can vary with the intended use of the findings, managers also require flexibility in evaluation systems. Table ES.1 summarizes various levels of rigor required in eight water quality management programs where the SI process can be applied.

Table ES.1. Summary of the use of Stressor Identification (SI) in water quality management programs.

Water Program	Type of Program			Level of Rigor Needed for SI			
	Advisory	Regulatory	Enforcement	Low	Medium	High	ID Source
305(b) Water Quality Reports	✔			✔	✔		✔
303(d) Impaired Waterbody Lists		✔				✔	✔
319 Non-point Source Control	✔			✔	✔		✔
402 Point Source Permitting		✔	✔			✔	✔
316(b) Cooling Water Intake Permitting		✔	✔		✔		✔
401 Water Quality Certifications		✔			✔		
404 Wetlands Permitting		✔	✔		✔		✔
Water Enforcement			✔			✔	✔

ES.4 Document Overview

The SI guidance document describes the organization and analysis of available evidence to determine the cause of biological impairment. The document does not directly address biological assessment, impairment detection, source allocation, management actions, or data collection, although these activities interact with SI in significant ways. This document is intended to guide water resource managers through the Stressor Identification process.

Section One: ***The Stressor Identification Process***
Introduces SI process and provides detailed guidance on implementing a stressor identification program. The guidance applies principles of ecoepidemiology to evaluating causes of biological impairment at specific locations.

Chapter 1: ***Introduction to the SI Process***
Provides the background and justification for the SI process.

Chapter 2: ***Listing Candidate Causes***
Provides an overview of and guidance on the first step of the SI process, listing candidate causes for the impairment.

Chapter 3: ***Analyzing Evidence***
Provides an overview of and guidance on the second step of the SI process, analyzing new and previously existing data to generate evidence.

Chapter 4: ***Characterizing Causes***
Provides an overview of and guidance on the third step of the SI process, using the evidence from Step 2 to draw conclusions about the stressors that are most likely to have caused the impairment.

Chapter 5: ***Iteration Options***
Provides options for stressor identification if no clear cause is found in the first iteration.

Section Two: ***Case Studies***
Provides two case studies illustrating the SI process.

Chapter 6: ***Presumpscot River, Maine***

Chapter 7: ***Little Scioto River, Ohio***

Appendix A: ***Overview of Water Management Programs Supported by the SI Process***

Appendix B: ***Worksheet Model***

Appendix C: ***Glossary of Terms***

Appendix D: ***Literature Cited***

Chapter 1:

Introduction to the Stressor Identification (SI) Process

In this Chapter:

1.1 Introduction

The use of biological assessments and biocriteria in state and tribal water quality standards programs is a top priority of the U.S. Environmental Protection Agency (EPA). As such, one of the agency's objectives is to ensure that all States and Tribes develop water quality standards and programs that

- use bioassessment information to evaluate the condition of aquatic life in all waterbodies,
- establish biologically-based aquatic life use designations,
- protect aquatic life use standards with narrative or numeric biocriteria (see box below),
- regulate pollution sources,
- assess the effectiveness of water quality management efforts, and
- communicate the condition of their waters.

SI is an invaluable component of any bioassessment/biocriteria program concerned with protecting the biological integrity of aquatic ecosystems.

Although bioassessments are useful for identifying biological impairments, they do not identify the causes of impairments. Linking biological effects with their causes is particularly complex when multiple stressors impact a waterbody. Investigation procedures are needed that can successfully identify the stressor(s) and lead to appropriate corrective measures through habitat restoration, point and non-point source controls, or invasive species control. Water management programs have historically shown that aquatic life protection is best accomplished using integrated information from various sources. For example, the whole effluent toxicity program has utilized methods for more than a decade that help resource managers understand and control the toxicity

Defining Terms– *Aquatic Life Use* and *Biocriteria*

Aquatic Life Use is a beneficial use designation, identified by a state, in which a waterbody provides suitable habitat for the survival and reproduction of desirable fish, shellfish, and other aquatic organisms. ***Beneficial Use Designation*** is a management objective defining desirable uses that water quality should support. Examples include drinking water supply, primary contact recreation (swimming), and aquatic life use.

Biocriteria are narrative expressions (qualitative) or numeric values (quantitative) describing the biological characteristics of aquatic communities based on appropriate reference conditions.

of complex effluents. Similarly, the Stressor Identification process will enable water resource managers to better understand and control stressors affecting aquatic biota. SI is an invaluable component of any bioassessment/biocriteria program concerned with protecting the biological integrity of aquatic ecosystems.

1.2 Scope of this Guidance

The SI guidance covers the organization and analysis of available evidence to determine the cause of biological impairment. It does not directly address biological assessment, reference condition, impairment detection, source allocation, management actions, data collection, or stakeholder involvement– although these activities interact with SI in significant ways. After stressors are identified, the appropriate management actions depend on the nature of those stressors, and on other factors– including economics. Identifying appropriate management actions is beyond the scope of this document, but examples of management actions are included in the case studies described in Chapters 6-7 of this document.

The SI process may be applied to any level of biological organization (e.g., individuals, populations, communities) and to any type of waterbody (e.g., freshwater streams, estuaries, wetlands, etc.).

Many methods exist for measuring impacts, exposure, land-use, habitat changes and other parameters that are important pieces of evidence in an SI investigation. Descriptions of those methods are beyond the scope of this guidance. The SI guidance, however, relies on the proper use of many tools to collect evidence. EPA recognizes the need for a tools compendium as well as software to help organize evidence, to make use of available databases and technical publications and to prompt proper collection of additional data when needed. The SI process should be viewed as a "logic backbone" in determining the cause of impacts to aquatic biota.

1.3 Data Quality Issues

The SI process is a procedure for analyzing available evidence and determining if the available evidence is adequate to draw a conclusion about the causes of impairment. Since evidence may be collected from a variety of sources using a variety of tools, proper documentation of the data is critical. Each technique for collecting data has associated quality control measures. The higher the quality of data analyzed, the better the chances will be of correctly identifying stressors. Guidance on assessing data quality and making use of various types of data may be found in the Comprehensive State Water Quality Assessment (305b) guidelines (USEPA 1997) and Ecological Risk Assessment guidelines (USEPA 1998a, also Chapter 3). Data of unknown or poor quality can sometimes be used for very rough estimates if the goals of the study allow, but, in general, the quality of all data should be acceptable and well documented. If the available data are not adequate, the SI process can show where data are missing or deficient, but it does not address designing new data collection efforts. Chapter 2, however, does provide advice on quality control when new data are collected.

After stressors are identified, the appropriate management actions depend on the nature of those stressors and on other factors, including economics. Evaluating whether stressor controls have allowed biological recovery is critically important in verifying that the stressors were accurately identified.

1.4 Overview of the SI Process

The SI process may be applied to any level of biological organization (e.g., individuals, populations, communities) and to any type of waterbody (e.g., freshwater streams, estuaries, wetlands, etc.). Some of the criteria presented for evaluating evidence may be specific, however, to a waterbody type (e.g., references to upstream/downstream associations). Similarly, the logic of the SI process may be applied in straightforward, single stressor situations or in complex situations with multiple stressors and cumulative impacts. Complex situations may require investigators to refine the definition of the study area, gather new data, or do multiple iterations of SI to identify all the important stressors. The Little Scioto Case Study (Chapter 7) is given as an example of a complex stressor situation where river segments were analyzed separately because impacts and stressors differed at each location.

1.4.1 The SI Process

Figure 1-1 provides an overview of the Stressor Identification process within the context of water quality management and data collection. The SI process is initiated by the observation of a biological impairment (shown in the topmost box). Decision-maker and stakeholder involvement is shown along the left-hand side; their involvement is particularly important in defining the scope of the investigation and listing candidate causes. At any point in the process of identifying stressors, a need for additional data may be identified; the acquisition of this data is shown by the box on the right-hand side of the diagram. The accurate characterization of the probable cause allows managers to identify appropriate management action to restore or protect biological condition. Once stressors are identified and management actions are in place to control them, the effectiveness of the SI process (as demonstrated by improved conditions) can be monitored using appropriate monitoring tools and designs.

The core of the SI process is shown within the bold line of Figure 1-1 and consists of three main steps:

1. listing candidate causes of impairment (Chapter 2),

2. analyzing new and previously existing data to generate evidence for each candidate cause (Chapter 3), and

3. producing a causal characterization using the evidence generated in Step 2 to draw conclusions about the stressors that are most likely to have caused the impairment (Chapter 4).

The first step in the SI process is to develop a list of candidate causes, or stressors, that will be evaluated. This is accomplished by carefully describing the effect that is prompting the analysis (e.g., unexplained absence of brook trout) and gathering available information on the situation and potential causes. Evidence may come from the case at hand, other similar situations, or knowledge of biological processes or mechanisms. The outputs of this initial step are a list of candidate causes and a conceptual model that shows cause and effect relationships.

.

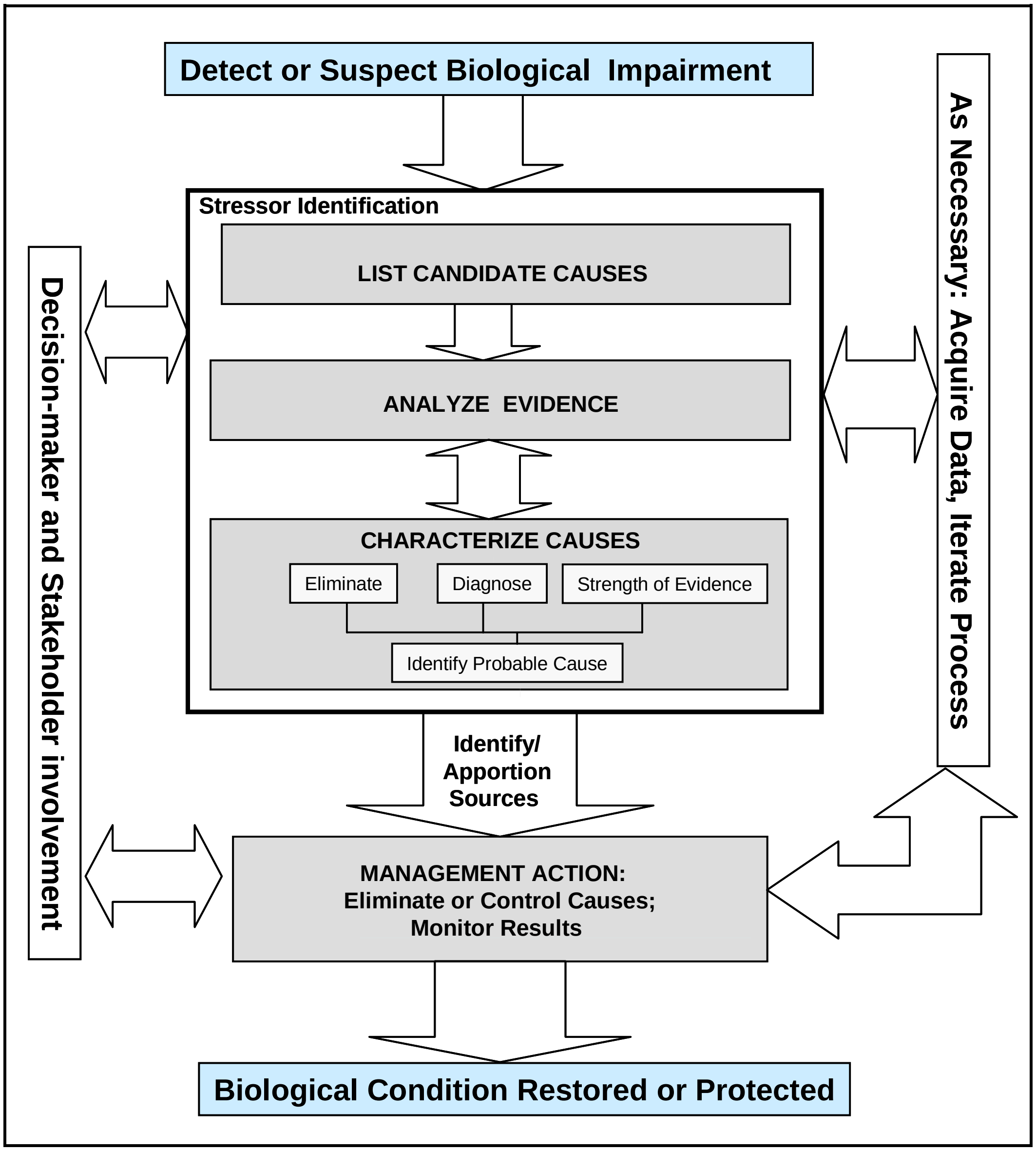


Figure 1-1. The management context of the SI process. (The SI process is shown in the center box with bold line. SI is initiated with the detection of a biological impairment. Decision-maker and stakeholder involvement is particularly important in defining the scope of the investigation and listing candidate causes. Data can be acquired at any time during the process. The accurate characterization of the probable cause allows managers to identify appropriate management action to restore or protect biological condition.)

The second step, analyzing evidence, involves analyzing the information related to each of the potential causes. Virtually everything that is known about an impaired aquatic ecosystem is potentially useful in this step. For example, useful data may come from chemical analysis of effluents, organisms, ambient waters, and sediments; toxicity tests of effluents, waters, and sediments; necropsies; biotic surveys; habitat analyses; hydrologic records; and biomarker analyses. These data do not in themselves, however, constitute evidence of causation. The investigator performing the analysis must organize the data in terms of associations that could support or refute proposed causal scenarios. Chapter 3 discusses several levels of associations between:

- measurements of the candidate causes and responses,
- measures of exposure at the site and measures of effects from laboratory studies
- site measurements and intermediate steps in a chain of causal processes, and
- cause and effect in deliberate manipulations of field situations or media.

These associations comprise the body of evidence used to characterize the cause.

In the third step, characterize causes, the investigator uses the evidence to eliminate, to diagnose, and to compare the strength of evidence in order to identify a probable cause. The input information includes a description of the effects to be explained, the set of potential causes, and the evidence relevant to the characterization. Evidence is brought in and analyzed as needed until sufficient confidence in the causal characterization is reached. In straightforward cases, the process may be completed in linear fashion. In more complex cases, the causal characterization may require additional data or analyses, and the investigator may iterate the process.

1.4.2 SI Process Iterations

The SI process may be iterative, beginning with retrospective analysis of available data. If the stressor is not adequately identified in the first attempt, the SI process continues using better data or testing other suspected stressors. The process repeats until the stressor is successfully identified. The certainty of the identification depends on the quality of information used in the SI process. In some cases, additional data collection may be necessary to confidently identify the stressor(s). Although the SI process cannot accurately identify stressors without adequate data, completing the SI process is helpful even without adequate data because the exercise can help target future data collection efforts.

Although the SI process cannot accurately identify stressors without adequate data, completing the SI process is helpful even without adequate data because the exercise can help target future data collection efforts.

1.4.3 Using the Results of Stressor Identification

Stressor Identification is only one of several activities required to improve and protect biological condition (Figure 1-1). In some cases, the most effective management action will be obvious after the probable cause has been identified. In many cases, however, the investigation must identify sources and apportion responsibility among them. This can be even more difficult than identifying the stress in the first place (e.g., quantifying the sources of sediment in a large watershed), and may require environmental process models. The identification and implementation of management alternatives can also be a complex process that requires additional analyses (e.g., economic comparisons, engineering feasibility) and stakeholder involvement. Once a management alternative is selected and implemented, monitoring its effectiveness can ensure that biological goals are attained, and provides valuable feedback to the SI process. All of these important activities are outside the scope of the current document. However, accurate and defensible identification of the cause through the SI process is the key component that directs management efforts towards solutions that have the best chance of improving biological condition.

1.5 Use of the SI Process in Water Quality Management Programs

Identifying the cause of biological impairments is an essential element of many water quality management programs. Table 1-1 summarizes the stressor identification needs of several water management programs. An extended discussion of some major regulatory programs and their requirements is presented in Appendix A.

Table 1-1. The role of SI in various water management programs.

Program Type/Name	Purpose	Role of SI
305(b) Characterizing the Quality of the Nation's Waters	Under section 305(b) of the Clean Water Act (CWA), states and tribes are required to assess the general status of their waterbodies and identify, in general terms, known or suspected causes of water quality impairments, including biological impairments.	Stressor identification procedures will assist states and tribes to accurately identify the causes of biological impairment. This is a non-regulatory, information reporting effort. A high degree of certainty in identifying the causes of impairment is not always needed for 305(b) reports.
303(d) Listings and TMDLs Identifying Waterbodies and Wetlands that Exceed Water Quality Standards	Under section 303(d) of the CWA, states and tribes are required to prepare and submit to EPA lists of specific waterbodies that currently violate, or have the potential to violate water quality standards, including designated uses and numeric or narrative criteria such as biocriteria. Wetlands assessment programs are also being developed and wetlands may be listed on 303(d) lists.	Accurate, reliable stressor identification procedures are necessary for EPA and the states/tribes to accurately identify the cause(s) of water quality standards violations. A high degree of accuracy and reliability in the stressor identification process is necessary and sources will need to be identified.
State/Local Watershed Management Programs	Managing water resources on a watershed basis involves examining the quality of a waterbody relative to all the stressors within its watershed. Stressors, once identified, are prioritized and controlled through a combination of voluntary and mandatory programs, possibly employing the CWA 402, 319, 404, 401, and other programs.	Stressor identification procedures will help to identify the different types of stressors within a watershed that may be contributing to biological impairment. A high degree of certainty in identifying the causes of impairment is needed.

Table 1-1 (continued). The role of SI in various water management programs.

Program Type/Name	Purpose	Role of SI
319 Non-point Source Control Program	The 319 Program is a voluntary, advisory program under which the states develop plans for controlling the impacts of non-point source runoff using guidance and information about different types of non-point source pollution.	Stressor identification procedures will help to identify the different types of non-point sources within a watershed that may be contributing to biological impairment. A high degree of certainty in identifying the causes of impairment is not always needed.
NPDES Permit Program	Under Section 402 of the CWA, it is illegal to discharge pollutants to waters of the United States from any "point source" (a discrete conveyance) unless authorized by a National Pollutant Discharge Elimination System permit issued by either the states or EPA. NPDES permits are required whenever a discharge is found to be causing a violation of water quality, including biological impairment.	Accurate stressor identification can be very critical in NPDES permitting cases, both for fairness and success in stressor control. The SI process can help to determine if the discharge is the cause of biological impairment. This is especially important when site-specific modifications of state standards or national criteria are used. A high degree of accuracy and reliability in the stressor identification process is necessary and sources will need to be identified. The SI process is not designed to allocate the amount of responsibility for an impact when multiple sources for a stressor are present.
316(b) Cooling Water Intake Program	Under Section 316(b) of the CWA, any NPDES permitted discharger which also intakes cooling water must not cause an adverse environmental impact to the waterbody.	To determine if a cooling water intake structure is causing adverse environmental impacts to the waterbody, the overall health of the waterbody should be known. Where biological impairments are found, stressor identification procedures should be used to identify the different stressors causing the waterbody to be impaired, including the intake structure. A high degree of certainty is needed.
401 Water Quality Certifications	Under Section 401 of the CWA, different types of federal permitting activities (such as wetlands dredge and fill permitting) require a certification that there will be no adverse impact on water quality as a result of the activity. This certification process is the 401 Water Quality Certification.	Stressor identification procedures will help to identify the different types of stress an activity may place on water quality that can then be addressed through conditions in the 401 Certification.

Table 1-1 (continued). The role of SI in various water management programs.

Program Type/Name	Purpose	Role of SI
Wetlands Permitting	Under Section 404 of the CWA, the discharge of dredge and fill materials into a wetland is illegal unless authorized by a 404 Permit. The 404 Permit must receive a 401 Water Quality Certification.	Stressor identification procedures may help to identify unanticipated stress from a dredge and fill activity on water quality or the biological community after the activity is underway. Stressor identification procedures will also help in pre-permitting evaluations of the potential impacts of 404 permitting by assessing different potential stressors on the wetland in advance.
Compliance and Enforcement	Whenever an enforcement action is taken by a regulatory authority, the type of pollution, the source, and other stressors that play a role in causing the violation need to be clearly identified and related to the violating source.	Stressor identification procedures must be able to clearly identify the different types of pollution causing the violation with a high degree of confidence. Legal defensibility is required. Identifying the source with a high degree of confidence is also needed, though the current SI process does not provide that guidance.
Risk Assessments	Results of bioassessment studies can be used in watershed ecological risk assessments to predict risk from specific stressors and anticipate the success of management actions.	Accurate stressor identification is an integral part of this process and can help ensure that management actions are properly targeted and efficient in producing the desired results.
Wetlands Assessments	States are beginning to develop wetlands assessment procedures. In the future, wetlands protection is expected to be increasingly incorporated into state water quality standards.	Stressor identification procedures, as well as future tools specific to wetland investigations, are very much needed by wetlands managers. The biological assessment methods will allow resource managers to evaluate the condition of wetlands and may provide some indication of the type of stressor damaging a wetland. Once bioassessment methods are completed and incorporated into monitoring programs, wetlands may be listed on 305(b) lists as impaired due to biological impairment. The SI process should help identify stressors causing biological impairment so resource managers can better remedy the problems.

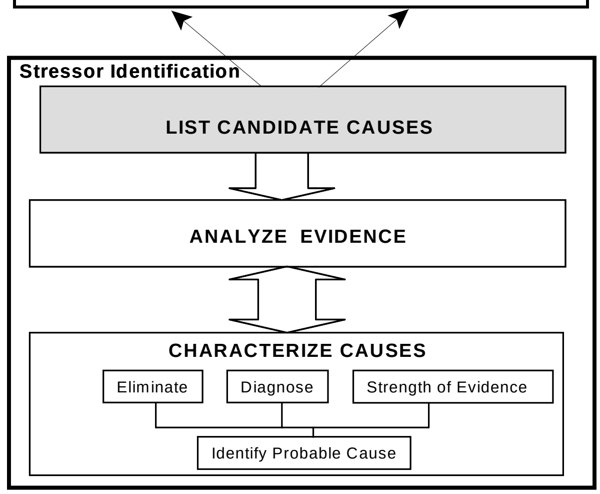
Table 1-1 (continued). The role of SI in various water management programs.

Program Type/Name	Purpose	Role of SI
Preservation Programs	The National Estuary Program (NEP) was established in 1987 by amendments to the Clean Water Act to identify, restore, and protect nationally significant estuaries of the United States. The program focuses on improving water quality in an estuary, and on maintaining the integrity of the whole system --its chemical, physical, and biological properties--as well as its economic, recreational, and aesthetic values.	Stressor identification procedures should be useful to the NEP, and other preservation programs, by helping stakeholders identify causes of impairments. This information would feed into the development of a management plan.
Restoration Programs	The Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), commonly known as Superfund, was enacted in 1980 (and amended in 1986) for hazardous waste cleanup.	As in enforcement and compliance programs, stressor identification procedures must be able to clearly identify the different types of pollution causing the impairment with a high degree of confidence. Legal defensibility is required. Identifying the source with a high degree of confidence is also needed, though the current SI process does not provide that guidance.
Pollution Control Effectiveness	A key component of any pollution control program or watershed management effort is the ability to ascertain (or predict) the likely effectiveness of pollution control measures or management strategies.	Stressor identification procedures will help to identify the different types of pollution a control measure needs to reduce and the different types of stressors a management strategy needs to address.

Chapter 2

Listing Candidate Causes

In this Chapter:



2.1 Introduction

The first step in the stressor identification process is to develop the list of candidate causes, or stressors. This is accomplished by carefully describing the effect that is prompting the analysis, and gathering available information on the situation and potential causes (see the box below for definitions of some key terms). Potential causes are evaluated and those that are sufficiently credible are retained as candidate causes used in the analysis stage. The outputs of this initial step are a list of candidate causes and a conceptual model that shows the relationship between the causes and the effect.

Defining Terms – *Exposures, Effects, Causes, Sources*

An ***effect*** is a biological change traceable to a cause.

Exposure is the co-occurrence or contact of a stressor with the biological resource.

A ***cause*** is defined as a stressor that occurs at an intensity, duration, and frequency of exposure that results in a change in the ecological condition.

A ***source*** is the origin of a stressor. It is an entity or action that releases or imposes a stressor into the waterbody.

note: the processes of detecting impairment and identifying sources are beyond the scope of this document

2.2 Describe the Impairment

The first important piece of information to be documented is a careful description of the effect that prompted the evaluation. Whenever possible, the impairment should be described in terms of its nature, magnitude, and spatial and temporal extent (see worksheet in Appendix B, Unit I, page B-4). Making inferences about causes is easier when the impairment is defined in terms of a specific effect, or response. The response should be quantified as a count (abundance of darter species) or continuous variable (mean length of darters). If multiple effects with different causes are described as a single impairment, it may be mistakenly assumed that there is only a single cause.

The importance of biological entities as resources and as sentinels of the overall integrity of ecosystems is recognized in the Clean Water Act as well as in subsequent legislation and regulations (See Chapter 1). Observations made in streams and rivers can alert environmental managers or the public to a potential problem. If the biological or ecological impairment is of sufficient magnitude, it may necessitate identifying the cause and the potential management controls needed to prevent further damage or to restore the ecosystem. Observations that might prompt the initiation of a stressor identification investigation include:

- kills of fish, invertebrates, plants, domestic animals, or wildlife,
- anomalies in any life form, such as tumors, lesions, parasites, disease,
- altered community structure such as the absence, reduction, or dominance of a particular taxon–this can include increased algal blooms, loss of mussels, increase of tolerant species, etc.,
- loss of species or shifts in abundance,
- response of indicators designed to monitor or detect biological, community, or ecological condition, such as the Index of Biotic Integrity (IBI) or the Invertebrate Community Index (ICI),
- changes in the reproductive cycle, population structure, or genetic similarity,
- alteration of ecosystem function, such as nutrient cycles, respiration, and photosynthetic rates, and
- alteration of the aerial extent and pattern of different ecosystems: for example, shrinking wetlands, change in the mosaic of open water, wet meadows, sandbars and riparian shrubs and trees.

It can be important to describe how the observed condition makes the waterbody unfit for its intended use. This makes the purpose and relative importance of the assessment clear. For instance, if the fish are covered with lesions, no one wants to fish for them.

In addition to describing the impairment, it is useful to prepare a background statement articulating the steps taken that revealed the biological impairment. For example, it might be appropriate to refer to a numerical or narrative biocriterion, or a reference condition that has been created for this type of waterbody, including the documentation for its derivation.

If conditions are below expectations, it is important to discuss how the quality or condition of the stream compares to other streams, or to the same stream in other places or times. Photographs of the water body provide visual evidence of a lost resource and can later be used in describing potential pathways that may have lead to the impairment. Equally important are photographs of what the resource could be like (e.g., taken from other locations), what it used to be like, or what valued attributes are still retained.

Maps or other geographical representations that show the location and severity of impairments are essential for orienting the investigators, examining spatial relationships, and eliciting information from stakeholders (see worksheet in Appendix B, Unit I, page

The scope of the investigation determines the extent of the data sets that will be analyzed. It defines the geographic area and time frame under consideration and the types of data that will be examined.

B-5). Maps can range from simple hand-drawn to computer-generated versions. Useful geographic information includes location of the impairment and known point sources, cities, roads, dams, tributaries, and land use. Examples of maps are included in the case studies presented in Section 2 of this document (Chapters 6 and 7). The depiction of this geographic information is also used to determine the scope of the evaluation; that is, the overall spatial and temporal extent of the study.

2.3 Define the Scope of the Investigation

The scope of the investigation influences the selection of candidate causes, and has ramifications for the final outcome and the practical use of the entire stressor identification effort. In a sense, the scope reflects perceptions about the ecosystem and beliefs about the level of restoration, or change, that is possible.

Early communication with the stakeholders will help ensure that relevant information has been identified and that potential causes are considered.

The scope of the investigation determines the extent of the data sets that will be analyzed (see worksheet in Appendix B, Unit I, page B-4). It defines the geographic area and time frame under consideration as well as the types of data that will be examined. The scope of the investigation may be limited or broad. An example of a limited scope is an evaluation of whether a particular stressor is responsible for an impairment. A broader objective would be to evaluate which, among several candidate causes, could be responsible for the observed effect. This broader approach might be appropriate for waters that are not attaining their designated use, and for which TMDLs (Total Maximum Daily Loads; see glossary) must be developed.

Several factors influence the overall scope of the investigation, including:

- the regulatory context,
- the purpose of the investigation,
- the relative importance of stressors emanating from outside the watershed,
- stakeholder expectations and interests,
- logistical constraints,
- cost,
- personnel, and
- available data.

Other factors to consider are the geographic extent of the impairment, and the extent of knowledge about the impairment. Early communication with the stakeholders will help ensure that relevant information has been identified, and that potential causes are considered. After these factors are carefully reviewed, a definition of the geographical area should be clearly stated. The regulatory context sometimes limits the scope of the study. For acid rain regulation, the geographical area is very large, whereas an NPDES violation may involve less than a kilometer of stream reach. The investigators should document any regulatory authorities involved and discuss the regulatory requirement for making a causal determination of the impairment.

The depth of the study may be limited by a paucity of data. In this case, it may still be appropriate to attempt a causal determination with the available data, and then indicate what additional information is needed to more confidently ascribe the cause.

2.4 Make the List

In developing a list of candidate causes, investigators should consider available evidence from the case at hand, other similar situations, and knowledge of biological processes or mechanisms (see text box entitled "Using Existing Programs to List Candidate Causes" and worksheet in Appendix B, Unit I, page B-6). The causes of ecological condition usually involve multiple spatial and temporal scales; both of which must be considered in defining the scope of the study and in listing candidate causes. Recent environmental events are overlaid on historical events, even those spanning geological time. Global and regional influences form the backdrop for local factors.

In some cases, two or more stressors must be present for the effect to occur.

Where multiple stressors contribute to cause an effect, the stressor that makes the largest contribution is the principal cause. Usually a principal cause is so dominant that removing other causes has no effect on the condition of the resource. For example, if benthic habitat is both physically altered and chemically contaminated, restoring the physical habitat may have no effect until the chemical contamination is removed. In this situation the chemical contamination is the principal cause. The habitat alteration is still a cause of impairment, but it is ancillary and masked by the toxic chemical impact. Nevertheless, pervasive ancillary causes like habitat alteration, nutrient enrichment, and sediment loading can lower the potential improvement to the waterbody even after the controlling or principal cause is removed.

Finding and Using Existing Lists of Stressors

Monitoring programs conducted by government agencies and non-governmental organizations may identify types and levels of stressors. For example, EPA's Environmental Monitoring and Assessment Program (EMAP) has monitored common stressors found in estuarine systems.[1] Among those listed are elevated nutrient concentrations, prolonged phytoplankton blooms, low dissolved oxygen, and sediment contamination.

State agencies and volunteer monitoring programs may also be good sources of information on stressors. Maryland's Department of Natural Resources (DNR), for example, maintains a website on which are links to maps indicating long term trends in total nitrogen, total phosphorus, and total suspended solids for 3rd order and larger streams in the state of Maryland.[2]

[1] See EPA "Condition of the Mid-Atlantic Estuaries." Office of Research and Development, Washington, D.C. #600-R-98-147. November, 1998.

[2] See Maryland DNR website, http://www.dnr.state.md.us/streams/status_trend/index.html

In some cases, two or more stressors must be present for the effect to occur. For example, a moderate level of nutrients poses no toxicological threat, but if sparse riparian cover permits sufficient sunlight to allow algal growth, then eutrophication can occur, with a subsequent cascade of effects. Another example is when a combination of reduced stream flow and lack of shading cause an elevation of temperature beyond the limit that native species can tolerate. Stressors acting together to cause an effect should be listed as a single scenario.

There are some ways to simplify the process of identifying and listing candidate causes. In the beginning, it helps to make a relatively long list and then pare the list down to the most likely causes. For the initial long list, it is a good idea to include all stressors known to occur in a waterbody. Even if these stressors have not previously been shown to cause this type of impairment, someone is likely to want proof that they were not causal agents. Include stressors that stakeholders have good reason to believe may be important. Consult other ecologists for potential causes of the impairment.

Knowledge about pollution sources near the waterbody can also suggest potential stressors. Point sources, such as drainage pipes, outfalls, and ditches are easily identified as sources. Constituents of the effluent can be listed as candidate causes. Other sources may be located some distance from the resource, such as motor vehicles and smoke stacks that generate candidate causes such as acid rain or nitrogen enrichment. Particular land uses often generate a consistent suite of stressors. For example, siltation and pesticides are commonly associated with agriculture. Locations of sources and stressors should be added to the impairment maps developed in Section 2.2.

Once an exhaustive list of candidate causes is developed, the next step is to pare the list down. Including very unlikely causes can make the identification process unwieldy and will distract stakeholders and managers from the more likely candidates. Unlikely stressors are those that are believed to be mechanistically implausible or absent from the watershed. Although they need not be evaluated, we recommend that you document the rationale for not including the less likely causes.

2.5 Develop Conceptual Models

The final part of this initial step is to develop conceptual models for the candidate causes, linking the cause with the effect (see worksheet in Appendix B, Unit I, page B-6). This part of the process documents a likely explanation of how the stressor could have caused the impairment. Conceptual models provide a good way to communicate hypotheses and assumptions about how and why effects are occurring. Models can also show where different causes may interact and where additional data collection may provide useful information.

The conceptual model can help the investigator see the pathway between the candidate cause and the eventual impact.

Conceptual models will vary in complexity, depending on the mechanisms and ecological processes involved. A generalized conceptual model might show land uses in the watershed that generate in-stream stressors impacting valued resources. For instance, if fish communities are impacted by moderate levels of nutrients in a sunlit stream, it is important to show that the effect could have occurred via several possible pathways, or a combination of pathways, such as:

- decaying algal blooms that result in low dissolved oxygen,
- the dominance of prey, causing a change in abundance of species,
- conditions favorable for opportunistic pathogens,
- diatom-rich water that is so turbid that sight-feeding fish cannot find prey and starve, and
- embedded substrates smothered with decaying and overgrown algal mats that reduce habitat for foraging, refugia, and reproduction.

The primary causes in this example are nutrients and incident sunlight. The secondary cause in the pathway could be any of the stressors that are formed from the initial cause. It is usually a good idea to consult with ecologists experienced with similar streams when developing conceptual models, especially when complex pathways and ecological process are involved.

Using a pictorial, poster-style conceptual model is useful to introduce the ecological relationships. Then a box and arrow diagram can be used to show details of the relationships among stressors, receptors, and intermediate processes. Some models get too complicated to be helpful. The diagram should show only the pathways and causes considered in the study. Separate diagrams for each stressor or pathway can keep the focus on the analysis steps that will follow. Figure 2-1 is an example of a box and arrow conceptual model illustrating the impacts of logging on salmon production in a forest stream. Additional examples and advice on conceptual model development can be found in Jorgensen (1994), Suter (1999), Cormier et al. (2000c), USEPA (1998a) (especially Appendix C), and in the case studies shown in Chapters 6 and 7.

In addition to helping the investigators to elucidate the relationships among multiple cause and multiple effects, conceptual models are also powerful tools for communicating among the investigative team and obtaining additional insights from stakeholders and managers.

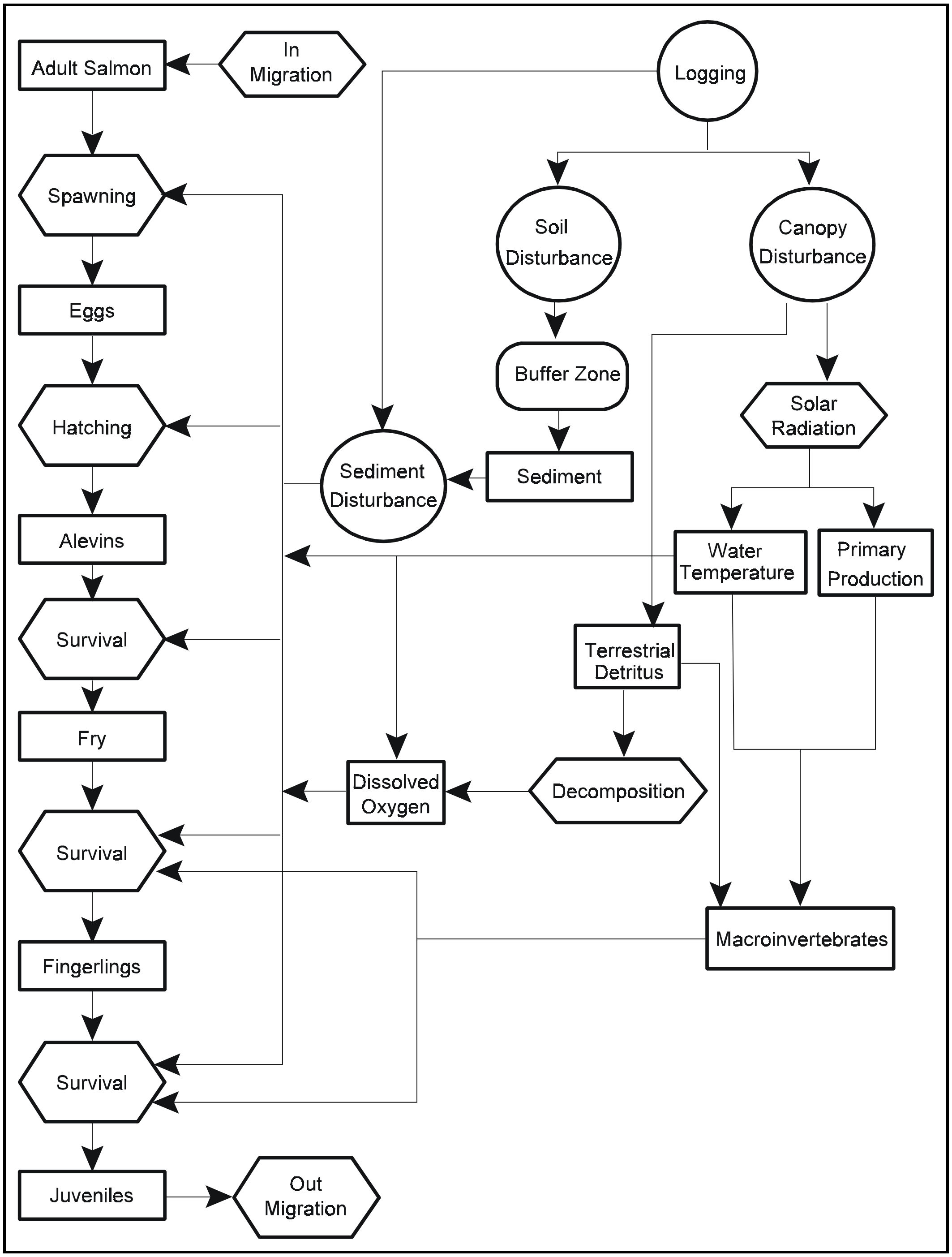


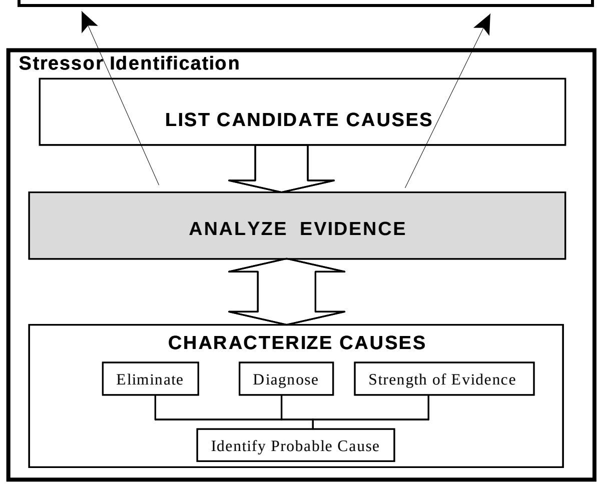
Figure 2-1. A conceptual model for ecological risk assessment illustrating the effect of logging in salmon production in a forest stream. (The assessment includes a series of exposures and responses. In the diagram, the circles are stressors, the rectangles are states of receptors, and the hexagons are processes of receptors. The rectangle with rounded corners is an intervention, establishment of buffer zones, that is being considered (Suter et al. 1994).)

Chapter 3

Analyzing the Evidence

In this Chapter:

3.1 Introduction
3.2 Associations Between Measurements of Candidate Causes and Effects
3.3 Using Effects Data from Elsewhere
3.4 Measurements Associated with the Causal Mechanism
3.5 Associations of Effects with Mitigation or Manipulation of Causes



3.1 Introduction

The second step in the SI process is to analyze the information that is related to each of the candidate causes identified in Chapter 2. Virtually everything that is known about an impaired aquatic ecosystem and about the candidate causes of the impairment may be useful for inferring causality. Potentially useful data that may come from studies of the site include chemical analysis of effluents, organisms, ambient waters, and sediments; toxicity tests of effluents, waters, and sediments; necropsies; biotic surveys; habitat analyses; hydrologic records; and biomarker analyses. A similar array of data may be obtained from other sites and from laboratory studies (performed *ad hoc* or reported in the literature). However, these data do not in themselves constitute evidence of causation. The investigators performing the causal analysis must organize and analyze the data in terms of associations that might support or refute proposed causal scenarios.

Existing data are often sufficient to determine the cause of impairment.

The SI process does not require a minimum data set, and existing data are often sufficient to determine the cause of impairment. However, the investigator has the responsibility of evaluating whether the data used are sufficient to support the SI process. If the investigator decides to generate additional data, its quality must be assured (see text box entitled "Data Quality Objectives").

The primary inputs to the analysis step are the list of candidate causes and the associated conceptual models that link the causes with the observed effects (developed in Chapter 2). Other inputs include data and information that come from the case at hand, other similar cases, the laboratory, and the literature that synthesizes biological and ecological knowledge (Figure 3-1). In the analysis step, this information is converted into causal evidence that falls into four general categories of relationships:

1. associations between measurements of the candidate causes and effects (Section 3.2),

2. associations between measures of exposure at the site and measures of effects from laboratory studies (Section 3.3),

3. associations of site measurements with intermediate steps in a chain of causal processes (Section 3.4), and

4. associations of cause and effect in deliberate manipulations of field situations or media (Section 3.5).

The evidence produced in the analysis step is used to characterize the cause or causes of the observed effect (see Chapter 4). The analysis and characterization of causes is usually done iteratively and interactively, as illustrated by the two-way arrows between the analysis and characterization boxes in Figures 1-1 and 3-1. Evidence is brought in and analyzed as needed until there is sufficient confidence in the causal characterization. In straightforward cases, the process may be completed in linear fashion. In more complex cases, the causal characterization may require additional data or analyses, and the investigator may repeat the process.

Data Quality Objectives

If new data will be generated for an SI investigation, consider following U.S. EPA's Data Quality Objectives (DQO) process. The DQO process combines a problem formulation exercise with conventional sampling statistics to determine the type, quantity, and quality of data needed to make an environmental decision with a desired probability of error (Quality Management Staff 1994). The DQO process is not directly applicable to SI since it is designed to determine the probability of exceeding a threshold. However, using a formal process to define the problem, examine information needs, and determine study boundaries is important in planning any sampling and analysis program. The criteria for defining an optimum design for an SI study will vary depending on the circumstances. Following sampling and analysis, a Data Quality Assessment (DQA) should be performed to determine whether the goals of the DQO process have been achieved (Quality Assurance Division 1998). The EPA's Quality System, including requirements for non-EPA organizations, can be found at www.epa.gov/quality/index.html.

Quality Assurance Division. 1998. Guidance for Data Quality Assessment. EPA QA/G-9, QA97 Version, or EPA/600/R-96/084. U.S. EPA, Washington, D.C.

Quality Management Staff. 1994. Guidance for the Data Quality Objectives Process. EPA QA/G-4, or EPA/600/R-96/055. U.S. EPA, Washington D.C.

3.2 Associations Between Measurements of Candidate Causes and Effects

The first type of evidence of causation is associations among measurements of candidate causes and effects (Table 3-1). The objective of this analysis is to provide evidence that:

- the candidate cause and the effect are observed at the same time or place,
- when the candidate cause is not observed, the effect is also not observed, or
- the intensity of the causal factor is related to the magnitude of the effect.

Causal evaluations often begin by examining associations from the case at hand. For example, effects are observed downstream, but not upstream of a candidate cause. These associations provide the core of information used for characterizing causes (see worksheet in Appendix B, Unit II, page B-7). Associations may be revealed by plotting data on common axes, as shown in Figure 3-2. In this figure, the spatial pattern of a toxicity bioassay results are clearly associated with the spatial pattern of a community metric. Causal inference is easier when the stressors and effects are located together (co-located) in time and space. Inference becomes more difficult as stressors are dispersed over larger scales, occur intermittently, or cannot be measured. Inference is also more difficult when there is a time lag between exposures. For example, if a stressor, such as a diversion of water flow prevents salmon from reaching the sea on their out-migration, the effect (i.e., destruction of the salmon run) may not be observed until three years later. In some cases, models may be useful for extrapolating inferences from available measurements.

Causal evaluations often begin by examining associations from the case at hand.

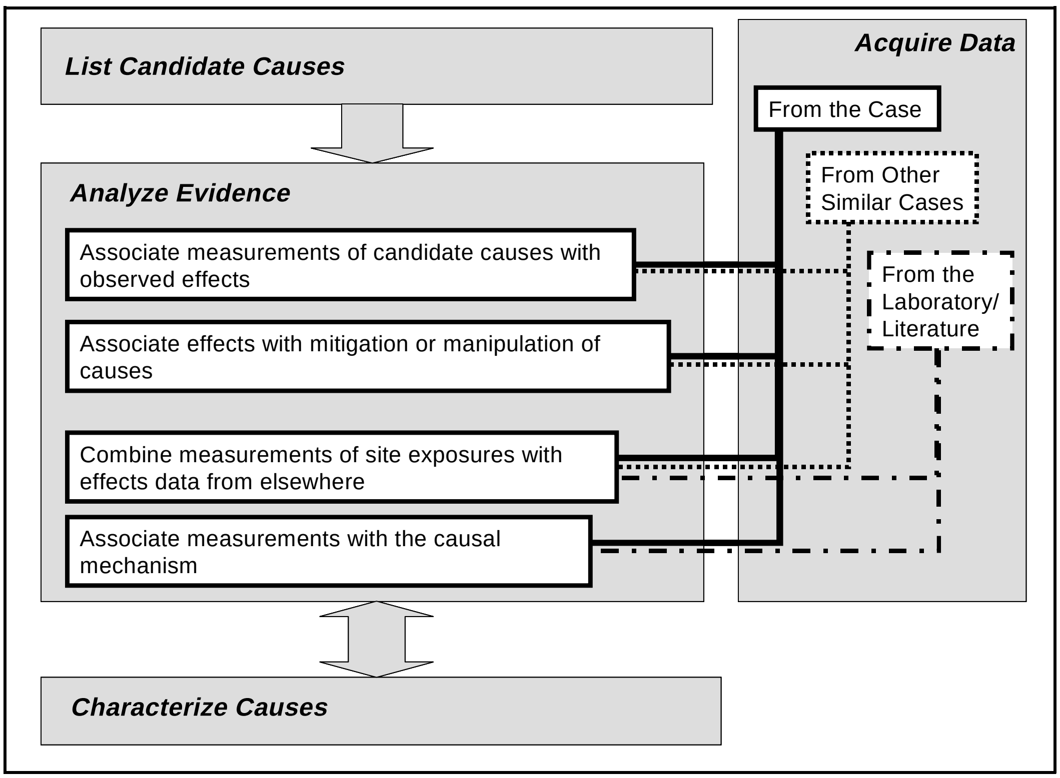


Figure 3-1. The flow of information from data acquisition to the analysis phase of the SI process.

Table 3-1. Types of associations between measurements of causes and effects among site data and the evidence that may be derived from each.

Type of Association	Example Evidence
Spatial co-location	Effects are occurring at same place as exposure Effects do not occur where there is no exposure For candidates with discrete sources on streams and rivers: Effects occur downstream of a source Effects do not occur upstream of a source For candidates with dispersed sources: Effects occur where there is exposure, but not at carefully matched reference sites where exposure does not occur
Spatial gradient	Effects decline as exposure declines over space
Temporal relationship	Exposure precedes effects in time Effects are occurring simultaneously with exposure (allowing for response and recovery rates) Intermittent sources are associated with intermittent exposure and effects
Temporal gradient	Effects increase or decline as exposure increases or declines over time

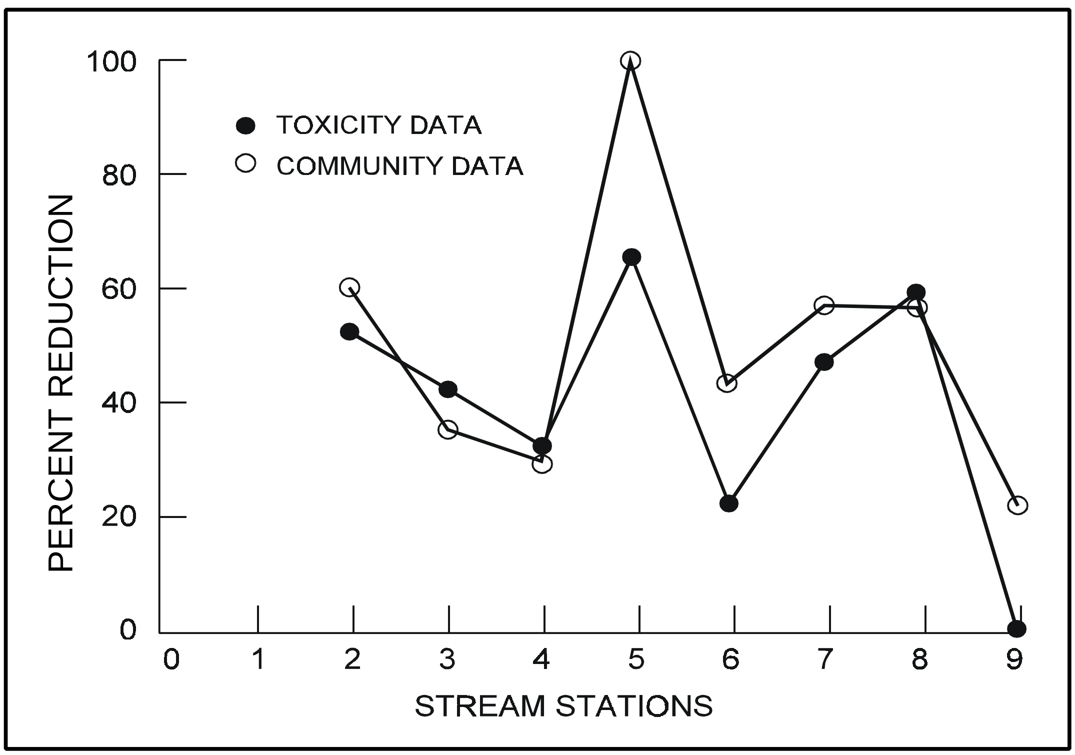


Figure 3-2. Plot of toxicity data from a 7-day subchronic test of ambient waters and a community metric obtained on a common stream gradient (Norberg-King and Mount 1986).

The evaluation of associations must consider whether potentially affected organisms may have moved since exposure. It is helpful to consider the mobility of organisms relative to the extent of the observed exposed and unexposed reaches or areas. Clearly, fish are capable of swimming long distances and invertebrates may drift downstream or fly upstream. However, extensive experience with bioassessment of fish and invertebrate communities has demonstrated that the movements of these organisms are usually not so great as to prevent the observation of spatial associations. The movement of a few individual organisms from contaminated reaches to upstream reaches will diminish, but generally not eliminate, the contrast or gradient among reaches. However, salmon and other species that regularly move long distances require special consideration when analyzing spatial associations. In such cases, consider the logic of the situation and possibly use a GIS as a platform for modeling spatial relationships.

Obtaining measurements of the stressor that can be associated with the effect can be challenging. In the most straightforward cases, the measurements of the stressor itself are available; for example, nutrient concentrations, degree of siltation, dissolved oxygen concentrations, or chemical concentrations. In some cases, the candidate cause is the lack of a required resource, such as nesting habitat. In these cases, measurements can establish that the resource is indeed missing at the place and time it would be required by an organism. When measurements of the stressor are not available, surrogates can be used, although the uncertainty in the analysis will increase. Information on the location and attributes of sources can be useful surrogates. This information can be particularly important for stressors that are intermittent in nature (e.g., high flow events), or degrade quickly (e.g., some pesticides). In these cases, source information may be used as a surrogate for the stressors. As sources become larger in scale and more diffuse, information on the sources becomes more difficult to use in site-specific causal evaluation.

In some cases, the candidate cause is the lack of a required resource, such as nesting habitat.

Similarly, measuring the immediate or direct response to a stressor increases the confidence in a causal evaluation. For example, a fish kill may be associated with nutrient enrichment, acting through algal growth, decomposition, and oxygen depletion. Measurements of the initial algal growth and oxygen depletion would increase an investigator's confidence that nutrient enrichment was the cause of the fish kill. Conceptual models are very useful for illustrating linkages between complex pathways of cause and effects, and for illustrating where measurements are (and are not) available.

Whenever possible, associations should be quantified. For categorical data, calculate the frequencies of associations. For count or continuous data use, linear or nonlinear models. For example, the abundance of Ephemeroptera at a site may be regressed against concentration of total sediment PAHs. Similarly, the community data plotted in Figure 3-2 might be regressed against the toxicity data. If effects data are categorical or heterogeneous and exposure data are continuous, categorical regression may be used (Dourson et al. 1997). Select the analysis technique that best illuminates the association, based on the amounts and types of data available. Some statistical descriptions of the associations include correlation coefficients, confidence intervals, and p-values. However, avoid statistical hypothesis testing of the associations (see text box entitled "Using Statistics and Statistical Hypothesis Testing for Analyzing Observational Data in Stressor Identification"). Because groups are not randomly assigned in a way that minimizes the influence of confounding variables, a significant outcome in a hypothesis test may be falsely attributed to a candidate cause, when in fact it is due to another factor. On the other hand, the small sample sizes that are usually seen in these studies decrease the ability to statistically discriminate groups, and may lead to mistakenly eliminating a true cause.

Often associations between candidate causes and effects can be improved by identifying and isolating confounding factors in either the receptors or the environment. For example, the frequency of hepatic neoplasms in fish is associated both with the age structure of the fish population and the concentration of PAHs in sediment (Baumann et al.1996). Correction for age of fish would increase the consistency and, potentially, the biological gradient in the relationship between hepatic neoplasm frequency and industrial contaminants. Similarly, a decline in fish species richness is a common measure of impairment, but the number of species present generally increases with increasing stream size (e.g., OEPA 1988a). Therefore, including a correction for stream size could strengthen the association between the degradation and species loss.

Associations observed from other studies can provide useful supporting information, particularly when the specific type or constellation of effects is consistently observed in association with a candidate stressor. Keep in mind that, as evidence, associations observed from other sites is not as strong as those observed from the study site. Therefore, if associations of effects and potential causes are analyzed at other sites, they should be evaluated separately from those at the site of concern.

3.3 Using Effects Data from Elsewhere

Measures of exposure from the case at hand can also be matched with measures of effect from other situations. The objective of this analysis is to provide evidence showing that the stressor is present at the study site in sufficient quantity or frequency that the investigator would expect to see a particular effect based on effect information from laboratory tests, field tests, or exposure-response relationships developed at other sites (see worksheet in Appendix B, Unit II, page B-12). This type of evidence is familiar to ecotoxicologists who combine measures of exposure from the study site with measures of effect from laboratory tests. For example, concentrations of chemicals measured in water may be compared to concentrations that are thresholds for effects in toxicity tests, or they may be used in concentration-response models to estimate the frequency or magnitude of effects. When doing these comparisons, the investigator should keep in mind that laboratory conditions or organisms may not accurately represent field conditions or organisms.

Equivalent measures of exposure and effects are available for non-chemical stressors (Table 3-2). As in toxicological assessments, it is important to choose the most applicable high-quality effect measurements. It is also important to ensure that the measures of exposure and effects are consistent. For example, long-term field exposures are most appropriately compared with chronic test data. In some cases, exposure-response information will not be available for a candidate cause, but will be available for an analogous agent, such as an effluent with a structurally similar chemical or an introduced species with similar feeding behavior.

Using Statistics and Statistical Hypothesis Testing for Analyzing Observational Data in Stressor Identification

Statistical techniques are essential tools for summarizing and analyzing environmental measurements for SI. Good SI uses a variety of techniques, including descriptive statistics (e.g., means, ranges, variances), exploratory statistics (e.g., multivariate correlations), statistical modeling (e.g, exposure-response relationships), quality assurance statistics (e.g., accuracy and precision of analyses of duplicates and standard reference materials) and comparison of alternative models of candidate causes (e.g., goodness-of-fit or maximum likelihood). However, the use of statistical hypothesis tests is problematic. Statistical hypothesis testing was designed for analyzing data from experiments, where treatments are replicated and randomly assigned to experimental units that are isolated from one another. The application of these tests to data from observational studies can result in erroneous conclusions. In observational studies, treatments are very seldom replicated and are never randomly assigned to experimental units.

If experimental units are replicated at all, they are replicated within the same water body and hence are likely to influence one another. As a result, samples are replicated rather than treatments. This is known as pseudoreplication (Hurlbert 1984). Finally, the location of a candidate cause is a given, rather than being randomly placed, so it is likely that candidate causes will co-vary with each other and with important natural attributes of the system (e.g., salinity, depth). The following table summarizes several common analytical techniques and discusses their use in SI.

Activity	Application to observational data in SI	Comments
Using summary statistics (e.g., mean water concentrations, 7Q10 flow rates) to summarize measurements	Encouraged	Pay attention to the biological or physical relevance of the summary statistic used. For example, the mean of chemical concentrations over time is often the most relevant (USEPA 1998a). As another example, the bankfull flow event is considered to be an important determinant of stream morphology (Rosgen 1996).
Using statistics to determine the probability that two sets or samples are drawn from the same distribution, or that they differ by a prescribed amount	Use Caution	Note that this use is not hypothesis testing in that it does not test a null hypothesis about a treatment (cause). It simply tells you the likelihood that differences are due to sampling variance. Also, the conventional criteria for statistically significant differences are not relevant; the differences must be shown to be biologically significant and the probabilities must be shown to affect the overall strength of evidence. Because the sample sizes are often small relative to variance, the power to detect real differences may be small.
Using the results of statistical hypothesis tests to conclude that a candidate is (or is not) the cause	Wrong	The assumptions of statistical hypothesis testing are violated. In observational studies, replicate treatments cannot be randomly assigned in a way that minimizes the influence of confounding variables. For this reason, a significant outcome in a hypothesis test may be falsely attributed to a candidate cause when in fact it is due to another factor.
Using correlations or regression techniques to quantify relationships between variables.	Encouraged	The type of data (continuous, ordinal, or categorical) and the type of relationship (e.g., linear, non linear) will determine the best technique to use.
Using statistics to determine the probability that a relationship is nonrandom, or that the slope of a regression differs from zero.	Use Caution	Note that this analysis indicates only the probability that an apparent relationship is due to sampling variance. It does not test the hypothesis that the relationship is causal. Also, the number of samples is likely to be low, so even correlations or models that are not statistically significant can be biologically significant and contribute to the strength of evidence.
Concluding that statistically significantly correlated variables have a causal relationship	Wrong	Correlation does not indicate causation, and a highly improbable regression model does not indicate that the independent variable caused the relationship. Because stressors often covary with each other and with natural environmental attributes, a strong relationship between a candidate cause and a biological variable may be due to a factor other than the candidate cause.

Table 3-2. Example associations between site-derived measures of exposure and measures of effects from controlled studies for different types of stressors.

Stressor	Characterization of Exposure: Intensity, Time, and Space	Characterization of Exposure-Response
Chemical	External concentration in medium Internal concentration in organism Biomarker	Concentration-response or time-response relationships from laboratory or other field studies
Effluent	Dilution of effluent	Effluent dilution - response in the laboratory (WET)
Contaminated Ambient Media	Location and time of collection Analysis of medium	Lab or *in situ* tests using the medium: Medium dilution - response Medium gradient - response
Habitat	Structural attributes	Empirical models (e.g., Habitat suitability models)
Water withdrawal/ drought	Hydrograph and associated summary statistics (e.g., 7Q10)	In-stream flow models (e.g., IFIM)
Thermal energy	Temperature	Thermal tolerances
Siltation (suspended)	Suspended concentration (e.g., TSS)	Concentration-Response relationships from laboratory or other field studies
Dissolved oxygen and oxygen-demanding contaminants (e.g. BOD, COD)	Dissolved Oxygen	Oxygen concentration-response relationships from laboratory or other field studies.
Siltation (bed load)	Degree of embeddedness, texture	Empirical siltation-response relationships from laboratory or other field studies.
Excess mineral nutrients	Dissolved concentration	Empirical concentration-response relationships from laboratory or other field studies. Eutrophication models
Pathogen	Presence or abundance of pathogen	Disease, Symptoms
Non-indigenous invasive species	Presence or abundance of the species	Ecological models (food web, energetics, predator-prey, etc.)

In developing mechanistic conceptual models depicting the induction of effects, it is often apparent that there are intermediate steps in the causal process that may be observed or measured.

Laboratory toxicity tests and other controlled studies provide the bases for models depicting the induction of effects by particular causes. For example, an acute lethality test of a chemical provides a concentration-response model which may be used to determine whether fish kills might be attributable to observed or estimated ambient concentrations. More complex causal mechanisms, particularly those involving indirect causation, require more complex mechanistic models. As models of causal processes become more complex, it becomes more difficult to judge whether an individual model provides an acceptable representation of the causes of ecological degradation at a site. In such cases, the best strategy is to generate mechanistic models of each proposed causal scenario and determine which model best explains the site data (Hilborn and Mangel 1997).

3.4 Measurements Associated with the Causal Mechanism

In developing mechanistic conceptual models depicting the induction of effects, it is often apparent that there are intermediate steps in the causal process that may be observed or measured. Documenting those intermediate steps increases confidence in the proposed causal mechanism (see worksheet in Appendix B, Unit II, page B-8). This type of evidence is particularly useful when the ultimate effects of multiple candidate causes are similar, but act through different mechanistic pathways. Types and examples of intermediate steps are presented in Table 3-3. In some cases it is sufficient to document the occurrence of the intermediate step, but in many cases, the level of the metric must be shown to be adequate. For example, if competition for prey by an introduced species is the proposed mechanism by which an endpoint species has been lost, then the investigator should show that the number of prey are reduced sufficiently.

Table 3-3. Example associations between site data and the processes by which stressors induce effects.

Type of Measurement	Example Mechanistic Association
Symptoms (i.e., responses specific to, or characteristic of, a type of stressor and causing the overt impairment)	Fish have lesions characteristic of a bacterium
Biomarkers	Metalothionine induction is an intermediate step in the glomerular toxicity of cadmium
Intermediate product of an ecological process	Algal abundance and DO are measures of intermediate steps in the induction of fish kill by nutrient additions
Changes in abundance of predators, prey, or competitors	Abundance of prey decreases upon introduction of a new predator

Table 3-3 (continued). Example associations between site data and processes by which stressors induce effects.

Type of Measurement	Example Mechanistic Association
Effects on other receptors	If impairment is defined in terms of effects on fish, then the responses of invertebrates or plants may suggest what causes are operating
Distributions of stressors and receptors coincide	For a stressor to cause an effect, it must contact or co-occur with the receptor organisms. For causes that act through the deprivation of a resource, the deprivation must actually occur

3.5 Associations of Effects with Mitigation or Manipulation of Causes

Strong causal evidence can be provided by deliberately eliminating or reducing a candidate cause and noting whether the effects disappear or remain (see worksheet in Appendix B, Unit II, page B-10). Causes can be eliminated as a part of a field experiment or by bringing site media into the laboratory (Table 3-4). Field experiments may also be performed by manipulating the source (see text box entitled "Associating Effects with Mitigation or Manipulation of a Cause"). For example, cattle may be fenced away from some locations where they usually have access to a stream channel, or an effluent may be eliminated for a time due to plant shut-down. These experiments may be conducted at the site being assessed, or may be conducted at other sites where the same type of source operates. Occasionally, a regulatory or remedial action may be treated as an experimental manipulation. Alternatively, experiments may be conducted that control the exposure of organisms or communities to potential causes. Examples include caging previously unexposed organisms at contaminated locations, placing containers of uncontaminated sediments in locations with contaminated water. These field experiments typically cannot be replicated, so their results are potentially subject to confounding (see text box "Using Statistics and Statistical Hypothesis Testing for Analyzing Observational Data in Stressor Identification"). Finally, site media can be brought into the laboratory and manipulated to eliminate different candidate causes. Then the results of the manipulation can be tested using laboratory organisms. These methods have been most extensively developed for the purpose of attributing causality among different chemicals in effluents.

Table 3-4. Types of field experiments and the evidence that may be derived from each.

Example Experiment	Example Evidence Derived from the Experiment
Manipulation of a source in the field	Elimination of a source reduces or eliminates the effect**.**
Manipulation of exposure in the field	Introduction of previously unexposed organisms results in effects. Isolation of organisms from one cause reveals the effects of others.
Laboratory manipulation and testing of media from the case	Extracting site media into fractions containing different chemical classes results in toxicity being associated with only one fraction.

Associating Effects with Mitigation or Manipulation of a Cause

Biological data collected by the Kansas Department of Health and Environment (KDHE) have played an increasingly important role in the state's efforts to document water quality impairments. KDHE historically has applied a modification of Davenport and Kelly's (1983) macroinvertebrate biotic index (MBI) to identify impairments resulting from nutrient loading and organic enrichment. Recently, a genus- and species-level indicator known as the Kansas Biotic Index (KBI) was developed to specifically respond to different stressor categories, including nutrients and oxygen demanding substances (KBI_{org}). Data collected by KDHE have shown that declines in the MBI and KBI_{org} have been consistently associated with increased organic enrichment, nutrient loading, and ammonia contamination.

The MBI and KBI_{org} were used to document the ***association between effects and the mitigation or manipulation of causes***. After a nitrification process was installed at the city of Wichita's municipal wastewater treatment facility, median concentrations of total ammonia-nitrogen in the Arkansas River decreased from 1.1 mg/L (1982-91) to 0.06 mg/L (1992-99). Concomitant decreases in the upper quartile MBI and KBI_{org} values were sufficiently large to justify a formal change in the Arkansas River's 305(b) impairment status. Moreover, city officials documented the recolonization of this river by several rare or previously extirpated fish species. Comparable improvements in MBI and KBI_{org} scores were documented in the Smoky Hill River below the city of Salina sewage treatment plant after ammonia levels were reduced by implementing wastewater nitrification and an industrial pretreatment initiative.

Outcome

In the 2000 KDHE 305(b) assessment, the Smoky Hill River was upgraded from non-supporting to fully supporting of aquatic life.

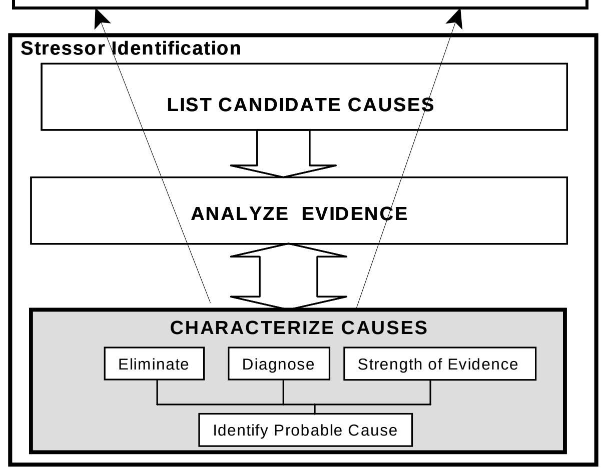
References

Davenport, E. and H. Kelly. (1983); Huggins, G. and F. Moffett. (1988); KDHE. (1993, 1998, 2000).

Chapter 4

Characterizing Causes

In this Chapter:



4.1 Introduction

Characterizing causes involves using the evidence analyzed in Chapter 3 to reach a conclusion and to state the levels of confidence in that conclusion. The input information in this process includes a description of the effects to be explained, the set of candidate causes developed in Chapter 2, and the causal evidence analyzed in Chapter 3.

4.2 Methods for Causal Characterization

After available evidence has been compiled and analyzed, the cause(s) may be obvious. In other cases, a more systematic method for reaching a conclusion may be needed. The use of clearly documented inferential logic increases the defensibility of causal attribution. This chapter describes three methods for using the evidence developed in Chapter 3 to characterize the cause: (1) eliminating alternatives, (2) using diagnostic protocols, and (3) weighing the strength of evidence supporting each candidate cause. Figure 4-1 depicts a procedure that combines these multiple methods to reach a conclusion of causality. Although this approach uses a combination of methods for characterizing causes, each method may also be used independently.

This integrated approach does not include all possible methods of causal analysis, particularly the use of expert judgment. When evidence is ambiguous, the process of developing consensus among a panel of experts may be more acceptable to stakeholders than any systematic evaluation of evidence. Utilizing expert judgment is certainly a more flexible approach in that it does not require any particular data set or type of model. In addition, experts can reach conclusions on the basis of experience and pattern recognition. For example, an experienced extension agent may visit a farm pond that is not producing bass and, without taking any measurements, know that the pond is too small or receives too much manure runoff from surrounding pastures to support bass reproduction. However, when the issue of causation is contentious, the attempt to develop consensus may be complicated by experts who represent the interests of the contending parties. Even when the experts are neutral, expert consensus may not be acceptable to some parties due to subjectivity. Finally, the process of developing expert consensus may not be practical. An NIH consensus development conference or an NRC panel may be practical for large-scale issues, such as the carcinogenicity of electromagnetic fields. It may not be practical to convene an expert panel for each outfall causing ecological injuries.

Although this approach uses a combination of methods for characterizing causes, each method may also be used independently.

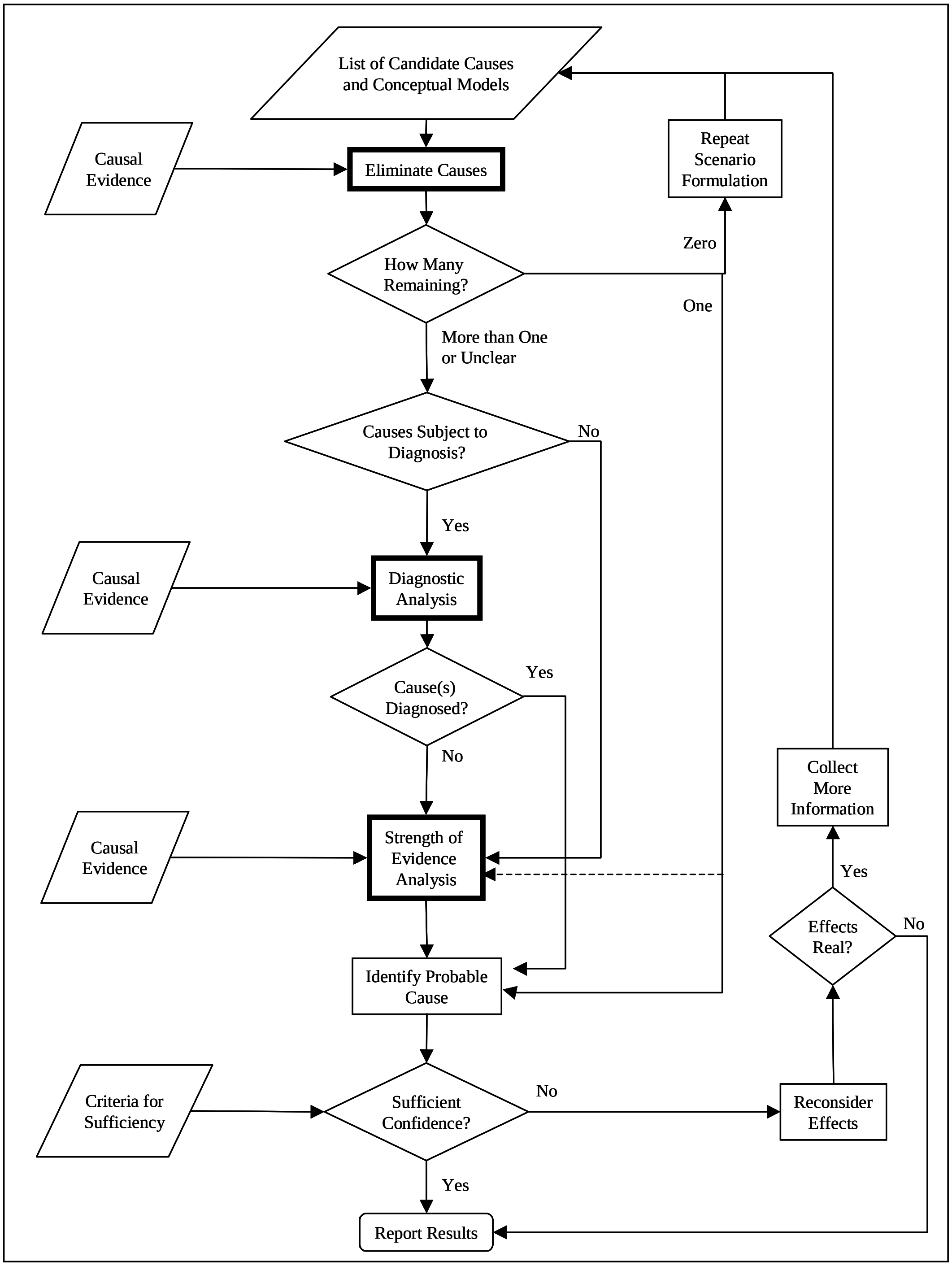
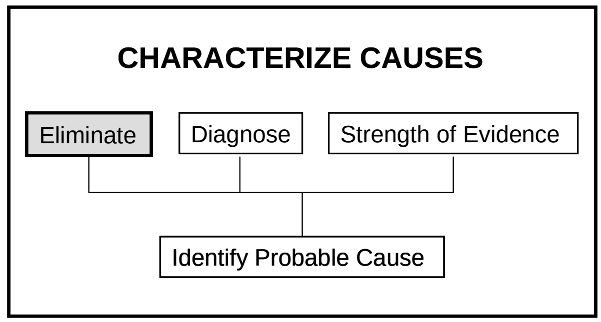


Figure 4-1. A logic for characterizing the causes of ecological injuries at specific sites. (Processes are rectangles, and the three inferential methods have heavy borders. Decisions are diamonds, and inputs are parallelograms.)

Inputs to the characterization process (the parallelogram at the top of Figure 4-1) include a description of the effects to be explained, the list of candidate causes, and the associated conceptual models (Chapter 2). The set of candidate causes should include stressors that consist of multiple factors that act together and are not individually sufficient to cause the effect (i.e., causal scenarios). Other inputs to characterization include the causal evidence produced in the analysis step (the three parallelograms on the left side of Figure 4-1). As discussed above, analyses are usually conducted in combination, as needed, throughout the characterization process. For example, the evidence necessary for eliminating candidate causes is analyzed first, then evidence for diagnosis, and, finally if necessary, the strength of evidence for each candidate cause is analyzed.

4.2.1 Eliminating Alternatives



The causal characterization methods shown in Figure 4-1 are presented in order, from the most conclusive to the least conclusive. The first method, eliminating alternatives, is a powerful approach to evaluating information. The ability to eliminate all but one alternative is a strong standard of proof for causality, and it is easily understood and widely practiced. It is the basic technique of literature's most famous master of inference, Sherlock Holmes:

> "When you have eliminated the impossible, whatever remains, however improbable, must be the truth."
> -- (Sir Arthur Conan Doyle, *Sign of Four*, 1890).

Elimination is also an effective way of reducing the numbers of alternatives to be considered before using another method (e.g., strength of evidence, Section 4.1.3, and see worksheet in Appendix B, Unit III, page B-15). Eliminating evidence is a particularly good option for SI when the set of alternatives is limited, and when disproof does not rely on statistics (see text box in Chapter 3 entitled "Using Statistics and Statistical Hypothesis Testing for Analyzing Observational Data in Stressor Identification"). Specifically, if the SI is conducted to support a permitting action, logical elimination of the permitted source as a potential cause of the observed injury is a sufficient causal analysis. Because of the complexity associated with ecological systems and multiple stressors, many SI investigations will not have the evidence necessary to confidently eliminate causes. These evaluations will rely on a strength of evidence analysis (Section 4.1.3).

Elimination as a method for establishing causality has strong roots in the philosophy of science. Popper, Platt, and other conventional philosophers of science have argued that it is logically impossible to prove a hypothesized relationship, but it is possible to disprove hypotheses (Platt 1964, Popper 1968). If a set of possible causes has been identified, once all but one alternative has been eliminated, the remaining hypothesis must be true. For example, if a body of water is found to be acidic, it is possible to establish the cause as acid deposition by eliminating acid mine drainage, geologic sulphate, and biogenic acids as causes (Thornton et al. 1994).

The elimination of alternatives has three major limitations:

- Due to limited knowledge, it may not be possible to identify a complete set of candidate. Also, the array of possible causes is potentially infinite, as there is no clear boundary between plausible and absurd hypothetical causes (Susser 1986b, Susser 1988).

- The process of elimination is limited by the ability to perform reliable tests and obtain unambiguous results. Such tests are often difficult in ecology. One may fail to reject a hypothesis but be uncertain of that result due to sampling variance, biases, and temporal variance. If all but one cause is rejected on uncertain grounds, it is difficult to accept the remaining candidate cause with confidence.

- Elimination of causes should be done with particular care when multiple sufficient causes may be operating. The evidence for one cause may be so strong that it masks the effects of another sufficient cause and appears to be the sole cause. In addition, beware that the temporal sequence of cause and effect may appear to be wrong when one sufficient cause precedes another. For example, an industrial effluent may impair a biological community. If the stream is subsequently channelized, the effects would be obscured by the industrial effluent. The channelization would have been sufficient to degrade biological communities within a pristine stream and therefore should be retained as a candidate cause. As shown in Table 4-1, similar issues are also relevant to spatial sequences such as those occurring in streams or rivers.

Most often the objective of SI is to identify all sufficient causes (for example, when the goal is to remediate or restore a water body). In these cases, the elimination step should be performed iteratively. That is, each cause eliminated during the first round should be reevaluated to determine if its effects may have been masked by another cause. If so, the candidate cause should be retained. In extreme cases, the masked secondary causes will remain unidentified, because the primary causes are so conspicuous. For example, if channelization has eliminated nearly all fish, it may not be apparent that episodic pesticide runoff would affect sensitive species. Such occult secondary causes will become apparent only after the primary causes have been remediated.

Some types of evidence can be used to eliminate candidate causes, and when those causes might be retained because of masking. Only associations derived from measurements taken from the case under evaluation are strong enough to eliminate an alternative. Associations derived from similar cases cannot be used to eliminate alternatives, but are useful in strength of evidence analyses which allow for uncertain or indecisive evidence (Section 4.1.3).

Only associations derived from measurements taken from the case under evaluation are strong enough to eliminate an alternative.

A stressor can be confidently eliminated if case-specific measurements clearly show that a necessary step in the causal chain of events has not occurred. For example, if a chemical must be taken up by an organism in order to cause an effect, and it can be demonstrated that uptake has not occurred (e.g., though biomarkers or body burdens), the chemical can be eliminated as a cause. Similarly, if sedimentation causes effects by silting-in riffles, and riffles can be demonstrated to be free of silt, sedimentation can be eliminated as a cause.

Although another potential way to eliminate a candidate cause is through experimental manipulations, the results of field experiments are seldom sufficiently conclusive to eliminate a cause. Uncertainties exist in field experiments due to a lack of thorough knowledge of recovery and recolonization rates following exposure. As a result, reduction or elimination of exposure may not appear to eliminate the effects. Field experiment data can, however, be used in the strength of evidence analysis discussed in Section 4.1.3. In addition, removal of one sufficient cause may unmask the effects of another. The protocols associated with the Toxicity Identification and Evaluation (TIE) program can be applied here, but not all effects of concern occur in these tests (e.g., tumors). Further, there may be questions concerning the sensitivity of the 7-day tests and test species relative to field durations and species (USEPA 1993a,b). TIE, therefore, is considered as part of the strength of evidence analysis.

Table 4-1. Application of common types of evidence in eliminating alternatives.

Type of Evidence (See Chapter 3)	Reason for Rejection	Masking Considerations	Causal Consideration [1] (See Section 4.1.3)
Associations between measurements of candidate causes and effects: Did the stressor precede the effect in time?	If the effects preceded a candidate cause in time, it cannot be the primary cause.	If the candidate cause is preceded by both the effect and another sufficient cause, its effects may be masked, and it should be retained.	Temporality
Associations between measurements of candidate causes and effects: Is there an upstream/ downstream conjunction of candidate cause and effect?	If the effect occurs upstream of the candidate cause's source or does not occur regularly downstream (e.g., is distributed spatially independently of a plume, sediment deposition areas, etc.), it cannot be the primary cause.	If the candidate cause is downstream of another sufficient cause, its effects may be masked and it should be retained.	Co-occurrence
Associations between measurements of candidate causes and effects: Is there a reference site/test site conjunction of candidate cause and effect?	If a candidate cause occurs at reference sites and occurs at equal or greater levels, it can be eliminated.		Co-occurrence

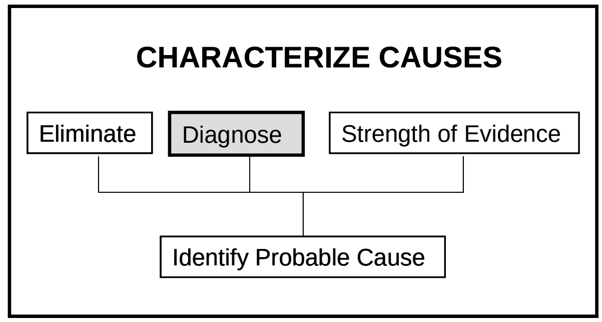
Table 4-1 (continued). Application of common types of evidence in eliminating Alternatives.

Type of Evidence (See Chapter 3)	Reason for Rejection	Masking Considerations	Causal Consideration [1] (See Section 4.1.3)
Associations between measurements of candidate causes and effects: Is a decrease in the magnitude or proportion of an effect seen along a decreasing gradient of the stressor?	A constant or increasing level of effect with significantly decreasing exposure would eliminate a cause.	If a decreasing gradient of one sufficient cause coincides with an increasing gradient of second, recovery from the first cause may be obscured.	Biological Gradient
Measurements associated with the causal mechanism: Has the stressor co-occurred with, contacted, or entered the receptor(s) showing the effect?	If the candidate cause never contacted or co-occurred with the receptor organisms, the cause may be eliminated. For appropriate stressors, if tissue burdens or other measures of exposure are found not to occur in affected organisms, the cause may be eliminated. For stressors that act through a known chain of events, if a link in the chain can be shown to be missing, the candidate cause can be eliminated.		Complete Exposure Pathway
Association of effects with mitigation or manipulation of causes: Did effects continue when a source or stressor was removed?	If the effect continues even after the stressor is removed, then the candidate cause can be eliminated. This assumes that there is no impediment to recolonization.	The effect may also continue if another sufficient cause is present.	Experiment, Temporality

[1] Many of the same types of evidence can also be used in the strength of evidence analysis (see Section 4.1.3). This column denotes the corresponding causal consideration used there.

In some cases all causes but one will be eliminated, and the part of the process is to describe the level of confidence in the characterization. It is often desirable to perform a strength of evidence analysis of that cause to demonstrate that it is probable, given all available evidence. If the true cause was not identified as a candidate, it may be possible to eliminate all candidate causes. In that case, one must repeat the process of identifying candidate causes (Chapter 2). In most cases, the elimination of causes will simply narrow the set of candidates, which is always helpful. Then the process continues to the next step, which is the use of diagnostic protocols or keys.

4.2.2 Diagnostic Protocols or Keys



If more than one cause remains after the elimination step, the next step is to consider whether any of the causes are subject to a diagnostic analysis. Whereas the elimination step relies on negative evidence (e.g., an exposure pathway is not present), diagnostic protocols rely on positive evidence (e.g., a particular symptom *is* present). Diagnostic symptoms are also used in the strength of evidence analysis (under consistency of association and specificity; see Section 4.1.3). The diagnostic protocols referred to here have been used and tested sufficiently to be considered authoritative and some have been formalized into a set of rules or a key (e.g., Meyer and Barclay 1990).

In medicine, diagnostic protocols identify a disease by examining its signs and symptoms. The diagnostic process requires an understanding of mechanism, so most of the evidence comes from measurements associated with the causal mechanism (see Section 3.4 and worksheet in Appendix B, Unit III, page B-19). As in medical practice, diagnostic information in the SI process comes from the exposed organisms and includes symptomatology (i.e., signs of the action of the causal agent on the organisms), measures of internal exposure (e.g., isolation of pathogens or analysis of chemicals in organisms), or measurements of intermediate processes (e.g., a depressed pre-dawn dissolved oxygen level).

The diagnostic approach is a good alternative for SI when organisms are available for examination, when the candidate causes are familiar enough that they have made it into the protocols, and when there is a high degree of specificity in the cause, the effect, or both. As an example, protocols for the investigation of fish kills are particularly well established (e.g., Meyer and Barclay 1990) and consist of collection of site data concerning candidate causes (e.g., oxygen, pH, temperature, contaminant levels, and presence of toxic algae), site data concerning effects (e.g., taxa killed, duration of event, behavior of live fish), and necropsy results (e.g., lesions, pathogens, tissue contamination, or clinical signs such as blue stomach which indicates molybdenum toxicity). Meyer and Barclay (1990) even provide a dichotomous key for determining the causes of fish kills. Since an SI investigation is more likely to examine current biological

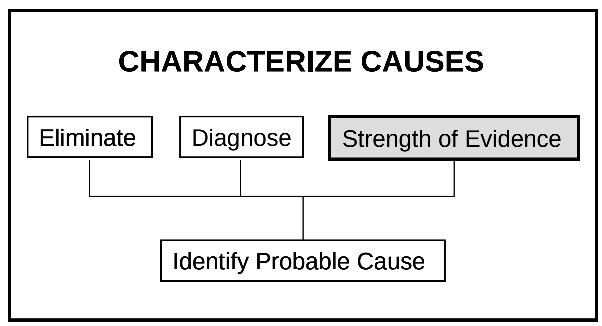
The diagnostic approach is a good alternative for SI when organisms are available for examination, when the candidate causes are familiar enough that they have made it into the protocols, and when there is a high degree of specificity in the cause, the effect, or both.

community compositions that might reflect past chronic exposures rather than the effects of acute lethality, the methods for fish kill investigations often are not directly applicable. However, a diagnostic approach can potentially be employed.

Diagnostic tools are well developed for pathogens and to a slightly lesser extent for chemicals (e.g., certain bill deformities are diagnostic of exposure to dioxin-like compounds) (Gilbertson et al. 1991). Diagnostics are also well developed for a few other agents such as low dissolved oxygen (low blood oxygen, gasping at the surface, etc.). For many other stressors and for most non-vertebrate aquatic organisms, reliable diagnostics are seldom available. Expert judgment has been used to assign tolerance values to taxonomic groups for nutrients and this concept has been extended to other stressor types (Hilsenhoff 1987, Huggins and Moffet 1988). The utility of using these tolerance values in multimetric indices along with some recent statistical analyses indicate that the structure of fish and invertebrate communities may prove valuable for diagnosis (Yoder and Rankin 1995b, Norton et al. 2000). Although the use of multimetric information for diagnosing cause and effect is not yet widely accepted or validated, this information can be brought into the strength of evidence analysis discussed in the next section.

4.2.3 Strength of Evidence Analysis

In many SI cases, the candidate causes are not identified by elimination or diagnosis, and an analysis of the strength of evidence for each of the candidate causes is required (see worksheet in Appendix B, Unit III, page B-20). This analysis organizes information so that the evidence that supports, or doesn't support, each candidate cause can be easily compared and communicated. When there are many candidate causes or when evidence is ambiguous, strength of evidence analysis is more useful than elimination of alternatives because it identifies the alternative that is best supported by the evidence. Even when a cause has been identified by a process of elimination or diagnosis, it is often desirable to complete the strength of evidence analysis in order to organize all of the evidence for the decision makers and stakeholders.



The strength of evidence analysis discussed in the remainder of this section defines a group of causal considerations used to organize the information concerning each alternative. Causal considerations are logical categories of evidence that are consistently applied to support or refute a hypothesized cause. They are defined in Section 4.2.3.1. Section 4.2.3.2 discusses how the types of evidence described in Chapter 3 provide information relevant to each consideration. Finally, Section 4.2.3.3 shows how to evaluate the strength of each piece of evidence in supporting or refuting a candidate cause.

For the purposes of this approach, we treat Koch's postulates (see text box entitled "Koch's Postulates") as a special case of analysis of the strength of evidence. That is, for pathogens or chemical contaminants, if Koch's postulates are satisfied, the strength of evidence is particularly high.

4.2.3.1 Causal Considerations for Strength of Evidence Analysis

This section describes various causal considerations used for strength of evidence analyses. These considerations draw on the work of epidemiologists and ecologists over the last 30 years (Fox 1991, Hill 1965, Susser 1986a).

Koch's Postulates

Koch's postulates combine different lines of evidence in a formal way to provide compelling evidence for causation. The approach was originally developed for pathogen-induced diseases. It has been adapted for demonstrating that particular toxicants cause human diseases (Yerushalmy and Palmer 1959, Hackney and Kinn 1979) or ecological effects (Adams 1963, Woodman and Cowling 1987, Suter 1990, Suter 1993), and has been recommended for ecological risk assessment (EPA 1998a). The following is an adaptation of Koch's postulates for causal inference in ecological epidemiology for effects of pathogens or chemicals.

1. The injury, dysfunction, or other potential effect of the pathogen or toxicant must be regularly associated with exposure to the pathogen or toxicant in association with any contributory causal factors.
2. The pathogen, toxicant, or a specific indicator of exposure must be found in the affected organisms.
3. The effects must be seen when healthy organisms are exposed to the pathogen or toxicant under controlled conditions, and any contributory factors should contribute in the same way during the controlled exposures.
4. The pathogen, toxicant, or a specific indicator of exposure must be found in the experimentally affected organisms.

The power of Koch's postulates arises from the way the four types of evidence are combined. The requirement of regular association in the field ensures that the association is relevant to the field, but, because field observations are uncontrolled, one cannot determine whether the association is, in fact, caused by an another agent that happens to be correlated with the proposed cause. In addition, associations in field data fail to demonstrate the temporal sequence between the candidate cause and effect. The requirement that the candidate causal agent induce the effect under controlled conditions eliminated the potential for confounding and demonstrates that the cause precedes the effect. However, the artificial conditions of toxicity tests and other experimental studies means that the demonstrated causal association may not be relevant to the field. The second and fourth postulates provide the ties that bind the two lines of evidence together. That is, evidence of exposure must be obtained in the field and must correspond to the experimental exposure. This correspondence of the exposure metrics makes it highly unlikely that the correspondence of effects in the field and the experiment are coincidental.

Koch's four postulates were derived for addressing the general issue of whether a stressor could be a cause at all (i.e., could DDT cause reproductive failure in birds). SI investigations typically choose among causal scenarios that have already been established as having the ability to produce impairment. For this reason, the emphasis is placed on postulate 2, identifying the pathogen, toxicant, or specific indicator of exposure in the affected organisms. This case-specific information is then combined with previously established information discussed in postulates 1, 3, and 4. This approach works best for simple causal agents that have a known indicator of exposure. When causal scenarios have multiple insufficient causes, the requirements of regular association and experimental evidence can rarely be met for the specific mixture that is encountered in the field situation. In cases where multiple sufficient causes can be assumed to be acting independently, the evidence for each cause can be evaluated separately.

The first four considerations, *co-occurrence, temporality, biological gradient*, and *complete exposure pathway* draw primarily on associations that are derived from the case itself. These considerations form the strongest basis for causal inference. The next two considerations, *consistency of association* and *experiment* can be based either on data from the case at hand or may draw from similar situations. The next four *considerations, plausibility, specificity, analogy, and predictive performance*, combine information from the case at hand with experiences from other cases or test situations, or from knowledge of biological, physical, and chemical mechanisms. These considerations provide corroborative information that can be used to supplement the basic observations of association of observed effects and potential causes from the case. The last two considerations, *consistency* and *coherency of evidence*, evaluate the relationships among all of the available lines of evidence.

Each of these causal considerations is discussed below:

Co-occurrence – The spatial co-location of the candidate cause and effect. In SI, this consideration is case-specific; for example, effects may be occurring downstream but not upstream of an identified source (see text box entitled "Arkansas River Case Study"). This consideration should be interpreted with caution when several sufficient causes may be present and when the objective of the analysis is to identify all potential and contributing causes. In this situation, the causes occurring the furthest upstream may mask the effects of causes occurring later in the downstream sequence.

Temporality – A cause must always precede its effects. For example, a baseline monitoring study showing a productive trout population before a dam was built provides some evidence that the dam caused the subsequent population decline. As with co-occurrence, this criterion should be applied with caution when several sufficient causes may be present and when the objective of the analysis is to identify all potential and contributing causes. In this situation, the causes occurring early in the time sequence may mask the effects of causes occurring later.

Biological Gradient – The effect should increase with increasing exposure. This is the classic toxicological requirements that effects must be shown to increase with dose. Biological gradient is also applicable to other types of causes (see text box entitled "Arkansas River Case Study"). For example, if fine substrate texture is believed to cause reduced diversity of benthic invertebrates, then diversity should decline along a gradient of texture. In SI, evidence for biological gradient is case-specific. Examples include demonstrating recovery of a community downstream of an outfall, or evidence that an effect decreases with decreasing concentration of an effluent or with increasing mean flow. Investigators should be aware that some stressors elicit non-linear response. For example, community diversity can increase at low levels of nutrient enrichment, then decline again as enrichment increases. Regression and correlation analyses are common tools used to quantify biological gradient; both high slopes and large correlation coefficients increase the strength of evidence.

Complete Exposure Pathway – The physical course a stressor takes from the source to the receptors (e.g., organisms or community) of interest. If the exposure pathway is incomplete, the stressor does not reach the receptor, and cannot cause an effect. Evidence for a complete exposure pathway is case-specific and may include measurements such as body burdens of chemicals, presence of parasites or pathogens, or biomarkers of exposure (see text box entitled "Arkansas River Case Study"). For stressors that do not leave internal evidence (e.g., siltation), measurements that show the stressor co-occurring in space and time with the receptor may be useful. For causes that induce effects indirectly, observations or measurements of the intermediate products or conditions are evidence of a complete exposure pathway (see Chapter 7, Little Scioto case study).

Consistency of Association – Refers to the repeated observation of the effect and candidate cause in different places or times (see text box entitled "Lake Washington Case Study"). A consistent association of an effect with a candidate cause is likely to indicate true causation. The case for causation is stronger if the number of instances of consistency is greater, if the systems in which consistency is observed are diverse, and if the methods of measurement are diverse. Consistency can be demonstrated using evidence from the case at hand, or may draw on evidence from many cases. For example, if fish kills repeatedly occur below a particular outfall, there is a consistent association over time of those incidents with a candidate cause. Less commonly, a particular case may have multiple instances of exposure to an agent spread over space rather than time. Consistent association can also be demonstrated across multiple sites or cases. For example, a decrease in benthic arthropod diversity may be consistently observed at many different sites having low dissolved oxygen levels. Consistency of association across many sites is seldom demonstrated because the same causal agent seldom occurs at multiple sites that are sufficiently similar to demonstrate a consistent response. However, when it is demonstrated, consistency across sites is stronger evidence for causation than the simple co-occurrence or temporal association of the agent with the response in a single case.

Arkansas River Case Study:
Using Strength of Evidence Analysis

This example highlights strength of evidence evaluations used in the SI process. Specifically, the example presents several lines of evidence used to support the hypothesis that heavy metal exposure impairs benthic macroinvertebrate communities.

Several sites in the Arkansas River (CO) were monitored over a 10-year span to examine the effects of cadmium (Cd), zinc (Zn), and copper (Cu) on benthic macroinvertebrates. More specifically, metal contamination was related to the abundance of heptageniid mayflies. It was found that heptageniid mayflies were abundant upstream of known metal inputs, and sparse downstream of these inputs, an example of ***spatial co-occurrence***. In addition, a ***complete exposure pathway*** was evident: concentrations of Cd, Cu and Zn were elevated in benthic invertebrates collected at stations downstream of the source. Evidence of a ***biological gradient*** was observed using multiple regression analysis; the abundance of heptageniid mayflies decreased with increasing zinc concentrations.

Evidence from other studies was also available and demonstrated that effects from metals would be ***plausible*** based on ***stressor-response*** relationships observed in the laboratory. Chronic toxicity tests of water collected from the Arkansas using *Ceriodaphnia dubia* and microcosm tests using mayflies established that effects would be expected at the concentrations of Zn, Cu, and Cd measured in the Arkansas.

Evidence from other studies also supported the hypothesis that heavy metal exposures reduce abundance of mayflies. Regional Environmental Monitoring and Assessment Program (R-EMAP) data from other locations in the Rocky Mountains showed a ***consistent association*** between metal exposures and reduced abundance of heptageniid mayflies.

Finally, efforts were undertaken by several agencies to reduce ambient metal concentrations, an example of a remedial ***experiment***. Increases in the abundance of heptageniid mayflies were observed at the sites with greatest metal reduction. Further, little biological improvement was observed where metal levels have remained elevated.

References: Clements and Kiffney 1994, Kiffney and Clements 1994a, Kiffney and Clements 1994b, Clements 1994, Clements et al. 2000, Nelson and Roline 1996.

Experiment – Refers to the manipulation of a cause by eliminating a source or altering exposure (Hill 1965) (see text boxes entitled “Lake Washington Case Study” and “Arkansas River Case Study”). Experiments of greatest relevance to SI (see Section 3.3) include manipulating and testing site media in the laboratory (e.g., using TIE), and conducting field experiments by controlling a source (e.g., fencing cattle) (USEPA 1991b, 1993a, 1993b). The strongest evidence is case-specific. If evidence from experiments conducted on a similar situation is used, the relevance to the case at hand should be described.

Plausibility – Refers to the degree to which a cause and effect relationship would be expected given known facts. Two types of plausibility are discussed below:

> <u>Mechanism</u>: Given what is known about the biology, physics, and chemistry of the candidate cause, the receiving environment, and the affected organisms, is it plausible that the effect resulted from the cause? It is important to distinguish a lack of information concerning a mechanism (e.g., the ability of chemical *x* to induce tumors is unknown) from evidence that a mechanism is implausible (e.g., chemical *x* is not tumorogenic). It is also important to carefully consider whether some indirect mechanism may be responsible. For example, increased nutrient levels cause algal blooms that decompose and reduce epibenthic oxygen concentrations, which in turn decrease invertebrate diversity. If a mechanism is known and there is evidence that the mechanism is operating in a specific case, the positive evidence is particularly strong.
>
> <u>Stressor-Response</u>: Given a known relationship between the candidate cause and the effect, would effects be expected at the level of stressor seen in the environment? The comparison of environmental concentrations to laboratory-derived concentration-response relationships is a common approach used in chemical risk assessments. It provides strong evidence of causality if concentrations are higher than a level that causes a relevant effect (see Table 3-2) (see text box entitled “Arkansas River Case Study”). Note that exceedence of water quality criteria or standards does not necessarily imply causation because regulatory values are intended to be set at safe levels. Whole effluent toxicity tests may be used with dilution models. Although used mostly for chemical stressors, a similar approach could also be used for other types of stressors, such as siltation.

Analogy – Examines whether the hypothesized relationship between cause and effect is similar to any well-established cases. Hill (1965) used the criterion of analogy to refer specifically to similar causes. For example, a new pesticide with a similar structure to another one may induce similar effects. The idea can be extended to other types of stressors. For example, an introduced species that has similar natural history characteristics to one that had been previously introduced may have similar impacts on the ecological system.

Lake Washington Case Study[1]

Lake Washington, located in Seattle and draining into Puget Sound, first began receiving street runoff and raw sewage input from Seattle at the turn of the 20th century. Although the sewer outlets were eventually replaced by wastewater treatment plant effluents, the growing human population in the surrounding area put increasing demands on the lake. By 1953, 10 wastewater treatment plants discharged into Lake Washington. Shortly thereafter, the first report describing nutrient loadings in the lake was issued by researchers at the University of Washington.

While the problems associated with eutrophication were not widely recognized by the public at the time, a University of Washington professor, W.T. Edmondson, used the concept of ***consistency of association*** to make an important observation: the recent discovery of a blue-green alga (*Oscillatoria rubescens*) in Lake Washington coincided with other documented cases where water quality had declined in response to nutrient input. The lakes described in these reports ranged geographically from Wisconsin to western Europe, yet the ***highly specific*** occurrence of *Oscillatoria* was identified in each case as an early response to water enrichment. Thus, Edmondson asserted that the water quality in Lake Washington was declining in response to nutrient input, and would continue to decline in predictable ways.

Edmondson developed a model based on principles of mass balance and stoichiometry to define the quantitative relationships between nutrient levels and algal biomass. He used the model to forecast that water quality in Lake Washington would continue to decline in predictable ways. This is an example of ***predictive performance***, since continued monitoring confirmed his assertions.

Outcome

Edmondson's letters and popular science articles describing the problems of the lake successfully brought about public and political support for the eventual clean-up of Lake Washington. Between 1963 and 1968, all 10 wastewater treatment plant discharges were diverted out of Lake Washington and sent to a common collection system that ultimately discharged deep within Puget Sound.

Until the diversions were constructed, water quality had continued to decline as predicted by Edmondson, with water transparency at less than 1 m in 1962. However, in the years following the improvements, nutrient levels decreased substantially. By the 1970s, visibility had reached 12 m, and the presence of the blue-green alga *O. rubescens* was undetectable. The swift recovery of Lake Washington following the removal of nutrient inputs in this field ***experiment*** left little uncertainty about the true cause of its water quality decline.

[1] Summarized from J. T. Lehman (1986).

Specificity of Cause – Applicable only if the proposed cause is plausible or if it has been consistently associated with the effect. Specific cause-effect relationships are more likely to be demonstrated to be causal (see text box "Lake Washington Case Study"). If an effect (e.g., hepatic tumors in fish) observed at the site has only one or a few known causes (e.g., PAHs), then the occurrence of one of those causes in association with the effect is strong evidence of causation. In the extreme, causation is clear when both effects and causes are specific (*x* causes specific effect *y*, and *y* is caused only by *x*). One implication of this consideration is that both effects and causes should be defined as specifically as possible in order to increase the specificity of the association. For example, a specific cause such as highly embedded substrate can be more clearly associated with identified effects than a general cause like overall poor habitat quality.

Predictive Performance – Refers to whether the candidate cause has any initially unobserved properties that were predicted to occur. Was that prediction confirmed at the site? The ability to make and confirm predictions is one of the hallmarks of a good scientific process. For example, if the proposed cause of a fish kill is drift of an organophosphate insecticide into a stream, one could make the specific prediction that cholinesterase levels would be reduced, or the more general prediction that insects and crustaceans would also be killed. If these predicted conditions are then observed at the site, it increases confidence in the causal relationship (see text box entitled "Lake Washington Case Study"). Multiple predictions in both the positive and negative direction would strengthen this criterion (e.g., plants and protozoa would not be harmed, but arthropods would be).

Consistency of Evidence – Refers to whether the hypothesized relationship between cause and effect is consistent with all available evidences. The strength of this consideration increases with the number of lines of evidence (Yerushalmy and Palmer 1959).

Coherence of Evidence – Examines whether a conceptual or mathematical model can explain any apparent inconsistencies among the lines of evidence. For example metal concentrations at the site may be sufficient to impair reproduction in fish, and yet both juvenile and adult fish occur at the site. This evidence may be coherent if reproduction is not occurring at the site, but juvenile fish re-colonize the site from unexposed locations. Another explanation may be that the measured total metal concentration is not 100% bioavailable. The strength of these explanations depend on the expertise and judgment of the assessors. It is a weak line of evidence, because of the possibility that *post hoc* explanations are wrong. However, the hypotheses may lead to experiments or predictions in future iterations of the causal assessment (e.g., testing the bioavailability of the metals), which could support stronger inferences.

4.2.3.2 Matching Evidence with Causal Considerations

Table 4-2 illustrates the different types of evidence discussed in Chapter 3 with the causal considerations they support. The relationship between types of evidence and causal considerations is not one-to-one. Each type of evidence may be relevant to several causal considerations, and a causal consideration may be evaluated using several different types of evidence. In any specific application of SI, evidence will probably exist for only some of the causal considerations, and the evidence will be uneven across the candidate causes. After the evidence relevant to each consideration is identified, it is evaluated as discussed in the next section.

4.2.3.3 Weighing Causal Considerations

Epidemiologists and ecoepidemiologists have attempted to develop guidance for weighing the causal considerations described below (Fox 1991, Hill 1965, Susser 1986a). Table 4-3 presents the possible outcomes for each consideration and provides symbols to represent the influence of each outcome on the inference.

Table 4-3 illustrates a format that can be applied to specific SI cases. In this table, the causal considerations are listed in the left-hand column. Each of the other columns presents results for a candidate cause. The rows show the appropriate number of +, -, or 0 symbols associated with the strength of evidence for each consideration evaluated for each candidate cause. Supporting narratives should describe how the scores were obtained from the evidence. We do <u>not</u> recommend adding up the scores for each candidate cause. Adding the scores erroneously implies that each consideration is of equal importance and is equitable only if the same types of evidence are available across all candidates. In difficult cases, it may be valuable to compare the evidence for each individual consideration across the candidate causes. Particular attention should be paid to negative results, which are more likely to be decisive.

Table 4-2. Types of evidence (columns) that contribute to each causal consideration (rows).

	Types of Evidence														
Causal Considerations	**Associations of Measurements of Cause and Effect**				**Measurements Related to Causal Mechanisms**					**Case Exposure/ Other Exposure Response**			**Experiments**		
	Spatial Co-location	Spatial Gradient	Temporal Co-occurrence	Temporal Gradient	Symptoms	Biomarkers	Indirect Effects (Abundances of other Species)	Effects on Other Receptors	Cause and Receptor are Co-located	Laboratory Exposure Response	Field-derived Exposure-response	Mechanistic model of Exposure-response	Manipulation of Sources	Manipulation of Exposure	Lab Manipulation and Testing of Media from Site
Co-occurrence	x														
Temporality			x										x		
Biological Gradient		x		x											
Consistency of Association	x		x												
Complete Exposure Pathway					x	x	x	x	x						
Specificity of Cause					x	x									
Plausibility: Mechanism					x	x	x	x	x			x			
Plausibility: Stressor-Response										x	x	x			
Experiment													x	x	x
Analogy	x	x	x	x						x	x	x			
Predictive Performance					x	x	x	x					x	x	

Table 4-3. Format for a table used to summarize results of an inference concerning causation in case-specific ecoepidemiology. (Table adapted from Susser (1986a), Fox (1991), Suter (1998), Beyers (1998).)

Consideration	Results	Score[1]
Case-Specific Considerations		
Co-occurrence	Compatible, Uncertain, Incompatible	+, 0, – – –
Temporality	Compatible, Uncertain, Incompatible	+, 0, – – –
Consistency of Association	Invariant, In many places and times, At background frequencies or many exceptions to the association	++, +, –
Biological Gradient	Strong and monotonic, Weak or other than monotonic, None, Clear association but wrong sign	+++, +, –, – – –
Complete Exposure Pathway	Evidence for all steps, Incomplete evidence, Ambiguous, Some steps missing or implausible	++, +, 0, –
Experiment	Experimental studies: Concordant, Ambiguous, Inconcordant	+++, 0, – – –
Considerations Based on Other Situations or Biological Knowledge		
Plausibility		
Mechanism	Actual Evidence, Plausible, Not known, Implausible	++, +, 0, –
Stressor-Response[2]	Quantitatively consistent, Concordant, Ambiguous, Inconcordant	+++, +, 0, –
Consistency of Association	Invariant, In most places, In some places, At background frequency or many exceptions to the association	+++, ++, +, –
Specificity of cause[3]	Only possible cause, One of a few, One of many	+++, ++, 0
Analogy Positive Negative	Analogous cases: Many or few but clear, Few or unclear	 ++, + – –, –
Experiment	Experimental studies: Concordant, Ambiguous, Inconcordant	+++, 0, – – –
Predictive Performance	Prediction: Confirmed specific or multiple, Confirmed general, Ambiguous, Failed	+++, ++, 0, – – –
Considerations Based on Multiple Lines of Evidence		
Consistency of Evidence	Evidence: All consistent, Most consistent, Multiple inconsistencies	+++, +, – – –
Coherence of Evidence	Evidence: Inconsistency explained by a credible mechanism, No known explanation	+, 0

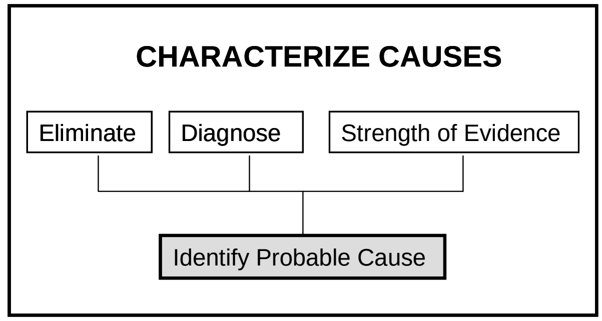
1 In addition to the scores noted, there ay be No Evidence (NE) available relevant to the consideration, or the consideration may be Not Applicable (NA) for the particular case (see especially stressor-response and specificity).

2 Stressor-response is not applicable (NA) if the mechanism is clearly implausible.

3 Specificity of cause is not applicable (NA) if either the mechanism is clearly implausible, or if there are many exceptions to the association.

Other methods for combining different lines of evidence include expert systems based on the logic of abduction and Bayesian statistical approaches (Josephson and Josephson 1996, Clemens 1986). As of this writing, these more quantitative approaches have not yet been developed for combining evidence for SI.

4.3 Identify Probable Cause and Evaluate Confidence



Whichever method is used to infer causation, the results of the characterization must be summarized. That is, the cause must be described, the logical basis for its determination summarized, and the uncertainties concerning that determination presented. As discussed above, there may be multiple sufficient causes, all of which should be characterized. In extreme cases, the effects of the primary causes are so severe that other potential causes will remain unidentified.

The level of confidence in causal identification may be assessed in quantitative or qualitative terms. Confidence is determined in part by uncertainty concerning the data, the models, and the observations that contribute to the inference. The uncertainty associated with the data may be partially estimated by conventional statistical analysis (see text box in Chapter 3 entitled "Data Quality Objectives," and "Using Statistics and Statistical Hypothesis Testing"), but also includes uncertainty concerning the applicability of the data. If data must be extrapolated between species or life stages, if old data are used to estimate current conditions, or if, for some other reason, data are not directly applicable, the associated uncertainty should be estimated. The uncertainty in statistical models, such as regressions of biological properties against levels of potential causes, may be estimated using goodness-of-fit statistics or confidence bounds. The uncertainty due to the parameters in mathematical models, such as models of dissolved oxygen depression due to nutrient input, may be estimated analytically or by Monte Carlo simulation (USEPA 1996a, 1999). If a causal inference is logically clear and is based predominantly on the results of a statistical or mathematical model, the uncertainties concerning the results may serve to estimate the uncertainties concerning the inference.

In most cases, unquantified uncertainties will dominate. These include lack of data concerning the presence or levels of particular stressors, incomplete biological data, uncertainty concerning the time when the impairment began, and many more. In addition, most causal inferences are based on the strength of evidence, so that no single source of uncertainty characterizes the uncertainty concerning the conclusion. Therefore, the uncertainty concerning most identifications of causes must be characterized qualitatively. That qualitative judgement should be accompanied by a list of major sources of uncertainty and their possible influence on the results.

In some cases, investigators will be able to clearly demonstrate that a particular cause is responsible for the ecological injuries of concern. However, in many if not most cases, there will be significant uncertainty concerning the relative contributions of alternative causal factors. In such cases, it is necessary to determine whether the evidence is sufficient to justify a management action. Standards and criteria for establishing epidemiological causation are not generally agreed upon. In particular, there is no consistent standard for adequacy of proof. While conventional science sets a high standard to prove causation, the precautionary principle begins by assuming that an agent is harmful and requires disproof of causation (Botti et al. 1996). Such decisions are made by risk managers, rather than risk assessors, and may be based on considerations such as the cost of remediation and the nature and magnitude of the ecological injury. Ideally, that judgment would be made on the basis of *a priori* criteria. That is, each program that uses SI should specify a standard basis for deciding whether the characterization of the cause is sufficient for the management purpose. For example, for the permitting of POTW effluents, a particular state might develop standards for proof that those effluents cause particular types of injuries. However, standards and criteria for establishing causation are not generally agreed upon, and many decisions are made *ad hoc*. That is, the evidence concerning causation may be presented to the risk manager as a best estimate of causation along with an accompanying analysis of uncertainties. The risk manager may use that result to help reach a decision.

As discussed in Chapter 1, the SI process may be conducted iteratively until sufficient confidence in the causal characterization is reached. In the most uncertain and complex cases, the SI process may best serve to guide further data collection, modeling, or analysis efforts. Options for iterating the process are discussed further in Chapter 5, below. If the cause is confidently identified, then the investigation may proceed to identifying sources, developing and implementing management options, and monitoring their effectiveness (Figure 1-1).

Chapter 5

Iteration Options

In this Chapter:

This chapter describes iterations if no clear cause can be identified. If the SI process has yielded no clear cause for the biological impairment, it may be because (1) there is actually no effect (Section 5.1), or (2) there is insufficient information concerning the identified causes or the true cause was not among the list of candidate causes (Section 5.2). These alternatives, all leading to a reiteration of the investigation (Figure 4-1), are discussed in this section.

5.1 Reconsider the Impairment

When no cause was identified, it may be that there is actually no effect, or the actual effect may be different from the identified impairment (see worksheet in Appendix B, Unit V, page B-35). This situation is known as a false positive, or in statistical terms, a Type I error. It should be noted that both false positive and false negative errors (failure to detect an effect that exists) are inherent to any detection system, whether it is medical diagnostics, aircraft radar, or environmental monitoring.

A false positive might result from errors in a biological survey or in the analysis of data. The samples may have been collected improperly; therefore, the biotic community appears to be less abundant or species rich than it truly is. The individuals performing the identifications may have misidentified organisms. There may have been errors in data recording or analysis. Any of these errors may artificially obscure the responses. A quality assurance program can minimize, but not entirely eliminate these errors. If the causal analysis reveals weaknesses in the evidence for the occurrence of a real effect, a careful audit of the biological survey may be appropriate.

Other reasons for a false positive result include sampling error and the natural variability of the biological indicators. In any monitoring program, sampling is stratified among perceived natural classes and subdivisions of systems (e.g., habitat type, salinity, sediment, elevation, biogeographic region), and often by season (sampling index period in defined season). A sample may have been taken outside of an index period. A site may belong to a poorly characterized system type or may have been incorrectly classified (e.g., cold water system evaluated using warm water criteria). Any unrecognized misclassification can result in either a false positive or false negative. Intensive monitoring and characterization of natural systems, combined with quality assurance and peer review of results, can reduce both types of errors.

In other cases, the impairment may have been defined too broadly or investigators may have made wrong assumptions about mechanisms when developing their conceptual model. For example, the first investigations into bird population declines and DDT focused on mortality rather than egg-shell thinning, and failed to find a connection with DDT (see text box entitled “Revisiting the Impairment in the Case of DDT”). Careful reconsideration of the nature of the impairment can put the investigation back on the right track.

Finally, natural variability of the indicators, not due to any measurement or analytical errors, can result in both false positives and false negatives. Environmental criteria may be defined by exceedence of a percentile or extreme value of some statistical distribution. This means that natural, or unimpaired conditions, may also exceed the criteria at some frequency. Ideally, acceptable error rates should be specified for decisions resulting from the biological assessment system. If confidence in a finding of biological impairment is low (that is, if the indicator just exceeds the threshold value), then increased sampling may reduce uncertainty and increase confidence (see next section).

5.2 Collect More Information on Previous and Additional Scenarios

If a causal scenario has not been established with sufficient confidence and the effect appears to be real, management should be consulted to discover if knowing the cause is still required for decision-making. If so, then more information must be collected (see worksheet in Appendix B, Unit VI, page B-36). Because the cost of field data collection and data analysis increases with each iteration, it is important to carefully plan what additional information is needed to determine the cause of impairment. This information may include previously considered scenarios for which information was inadequate, or candidate causal scenarios that were not previously considered.

Revisiting the Impairment in the Case of DDT[1]

The fact that DDT played a role in the decline of bald eagle and other bird-of-prey populations (e.g., osprey, brown pelicans) is now commonly appreciated among most biologists. However, the link between DDT and the eggshell thinning that caused reproductive failure in these birds was not initially recognized. Ultimately, the connection was made by re-examining the description of the impairment.

The first link between DDT and diminishing bald eagle and other bird-of-prey populations was the consistent observation of high body burdens of DDT metabolites. In other words, there was ***co-occurrence*** of the declining bird populations and the candidate cause, DDT. There was also evidence of ***a complete exposure pathway*** to birds based on body burden of DDT. However, extensive toxicity testing of DDT on adult bird mortality revealed no relationship. This suggested that the proposed mechanism, toxicity, was implausible. However, lethality was not the impairment; decline of birds-of prey was the impairment. A new ***conceptual model*** was required that considered other mechanisms that could result in declines in bird populations. In re-examination of the overall analysis, it became apparent that the species chosen for testing had been relatively tolerant of DDT exposure compared to those that were affected in the wild, and that the endpoint observed in these tests (lethality) would not reflect reproductive success or failure resulting from DDT exposure.

Field observations eventually revealed a potential ***plausible mechanism*** of reproductive failure due to eggshell thinning among bald eagles and other birds-of-prey. Laboratory ***experiments*** showed that DDE could cause eggshell thinning. Field studies showed that field exposures to DDE, a metabolite of DDT, were sufficient to cause effects in many species of birds based on the ***stressor-response*** relationship. Together these findings provided lines of evidence by which DDT might cause eggshell thinning and reduce reproductive success, a more specific impairment than declines in bird population.

Outcome

In 1972, DDT was banned from most uses in the United States. In the years following the ban, bald eagle and other bird-of-prey populations slowly recovered. The recovery of bird populations after banning the use of DDT, is an example of mitigation of the effect following manipulation of the cause, and is very strong evidence that the use of DDT was, in fact, the true cause of bald eagle and other bird-of-prey population declines.

References

Grier, J.W. 1982; Blus, L.J., and C.J. Henny, 1997

Even when the characterization of causes has not determined the cause with sufficient confidence, the set of candidate causes should have been reduced, and the critical evidence should be apparent. In particular, it should be possible to design experiments or observations that will potentially eliminate certain causes (Chapter 4.1.1). However, such experiments are not always feasible. Alternatively, one may identify critical pieces of positive evidence that would strongly support one scenario and none of the others. In most cases, it will be appropriate and prudent to plan a sampling and testing program that will generate a set of potentially decisive positive and negative evidence.

If all of the most common causes have been eliminated or have been determined to be unlikely, then additional causal scenarios need to be identified. The process is similar to that described in Chapter 2. New data may have come to light during the first iteration of the SI process. These data should be carefully reviewed to determine if there are any clues to suggest additional causal scenarios. Details of the available data should be considered, such as weather patterns, new construction, or land use information. If the descriptions of the effect or the scope were too broad, they may need to be refined or more clearly defined. Additional potential causal scenarios may include new stressors or combinations of stressors that occur simultaneously or in a specific sequence. After the additional candidate causal scenarios are developed, key evidence should be identified that is likely to allow identification of the cause.

The most important tools to bring to the SI process are experience in multiple disciplines (especially ecology), careful, deliberate critical thinking, and a strong desire to find the true cause of biological impairment.

Chapter 6

Presumpscot River, Maine

6.1 Executive Summary

The Presumpscot River is located in southern Maine and forms the outlet of one of Maine's largest lakes, Sebago Lake. From 1984 to 1996, biological monitoring downstream of a pulp and paper mill discharge consistently revealed non-attainment of Maine's Class C aquatic life standards. The river is impounded above and below the discharge. The discharge releases high concentrations of TSS and total phosphorus, and on occasion releases metals above the chronic criteria but below acute criteria. Upstream samples consistently indicated attainment of Class C or better standards.

<u>Description of the Impairment</u>

Biological impairment was characterized by a shift in the benthic macroinvertebrate community from 90% insects upstream of a pulp and paper mill discharge to about 50% insects downstream. This shift included a 15-35% loss of taxonomic richness, and 40-60% loss of Ephemeroptera-Plecoptera-Trichoptera (EPT) taxa. Moreover, many insect taxa found upstream of the discharge were pollution-sensitive, while those found downstream were primarily pollution-tolerant species, such as snails and worms.

<u>List Candidate Causes</u>

Eight candidate causes for non-attainment were considered in the Stressor Identification process:

1. Excess toxic chemicals from the discharge;
2. High TSS combined with floc causes high BOD and reduced DO;
3. High TSS combined with floc causes smothering;
4. Excess nutrients (from POTWs, nonpoint sources, and the mill) cause excess algal growth;
5. Impoundment increases sedimentation that smothers biota;
6. Impoundment decreases flow velocity and causes algal growth, leading to reduced DO;
7. Impoundment causes low DO; and
8. Impoundment causes loss of suitable habitat.

<u>Characterizing Causes: Eliminate</u>

Four of the eight candidate causes were logically eliminated from examination of the evidence. Reduced DO sufficient to cause the impairment was not observed in the Presumpscot River, and bottom-water DO concentrations were stable throughout the river, above and below the discharge. Therefore, causal scenarios #2, #6 and #7 could be eliminated. Although elevated concentrations of total phosphorus (TP) were observed below the discharge, the increase in chlorophyll *a* concentration was negligible. Water column chlorophyll *a* is a surrogate measure for algal biomass. Because excess algal growth was necessary for causal pathway #4, and there was none, it was also eliminated from further consideration.

<u>Characterizing Causes: Diagnose</u>

No evidence strong enough to support diagnosis was available for any of the candidate causes.

<u>Characterizing Causes: Strength of Evidence</u>

A strength of evidence approach was then used to examine the remaining four candidate causes. The four remaining causes were toxic chemicals, flocculent TSS causing smothering, impoundment increasing sedimentation, and impoundment causing loss of suitable habitat. There was no strong evidence for or against the toxic chemical hypothesis (#1). Several lines of strong evidence favored the TSS hypothesis (#3):

- The exposure pathway from discharge to biological impairment was complete and plausible.

- Other rivers with similar elevated flocculent TSS also had impaired biological assemblage.

- Removal of flocculent TSS from a nearby river resulted in recovery of the biological assemblage.

Two lines of evidence disfavored the two impoundment hypotheses (#5 and #8):

- Other impoundments with similar potential sediment loadings (not from mill discharge) and similar habitat support diverse invertebrate assemblages that meet aquatic life use criteria; and

- a site upstream of the mill effluent, and within the same impoundment, met aquatic life use criteria.

<u>Characterizing Causes: Identify Probable Cause</u>

The evidence supporting scenario #3, that non-attainment was due to high loads of flocculent TSS from the discharge, was consistent throughout the lines of evidence. Strength of association, spatial co-occurrence, and experimental lines of evidence strongly supported this scenario. Evidence for the toxicity scenario (#1) was extremely weak. Evidence for the two impoundment scenarios (#5 and #8) was negative. The State of Maine concluded that high TSS was sufficient for causing the biological impairment. Quality of the data were adequate, and confidence in the conclusion was high. Subsequently, the State took management action to reduce loadings of TSS through a TMDL that was approved by EPA. This was the first time in New England that bioassessment findings had served as the quantitative response variable for development of a TMDL and resulting pollutant discharge limits, including the pulp and paper mill.

Moreover, it provided a means for Maine to control a pollutant (TSS) for which it had no specific criterion in its water quality standards.

6.2 Background

This case study is presented as an example of how the stressor identification (SI) process could have been used to determine the cause of non-attainment of aquatic life use in a small river in Maine. The case study begins with the presentation of background information on the regulations in the State of Maine and the geographical location of the case study. This is followed by a brief discussion of the evidence found at the site and in other situations. Several causal scenarios are then presented and analyzed separately to illustrate how the SI process could be used to eliminate four of the eight candidate causes. A strength of evidence analysis is then used to identify the most likely cause. The case study concludes with a brief discussion of the management actions taken to remedy the situation. One of the most significant results of this effort was that the State of Maine, Department of Environmental Protection, used bioassessment findings to control a stressor for which the State has no standards.

Impairment Trigger: Biological monitoring in the Presumpscot River in Westbrook, below a pulp and paper mill discharge, has consistently revealed non-attainment of Class C aquatic life standards (1984, 1994, 1995, 1996) using standard Maine Department of Environmental Protection methods (invertebrate) (Davies and Tsomides 1997).

Regulatory Authority

The Maine Department of Environmental Protection (MDEP) issues wastewater discharge licenses that set the allowable amounts of pollutants that industries may discharge to waters of the State. These limits are scientifically determined in order to preserve water quality sufficient to maintain all designated uses and criteria established, by law, for the river. In recent years USEPA has required that a Total Maximum Daily Load (TMDL) be established for impaired river systems, such as the Presumpscot, for which existing, required pollution controls are inadequate to attain applicable water quality standards.

The State of Maine established minimum standards for three water quality classifications, Class A, Class B, and Class C. These classes specify designated aquatic life uses from Class C, the minimum state standard, to the most protected waters with the Class A/AA designation. Class C requires that the structure and function of the biological community be maintained and provides for the support of all indigenous fish species.

Under this system, attainment of the aquatic life classification standards for a given water body is evaluated using numeric biological criteria. The MDEP numeric aquatic life criteria are based on statewide data collections over a 14-year period with analysis of over 400 sampling events. Artificial substrates (rock baskets) are incubated on the bottom at stream sites, retrieved, and benthic macroinvertebrates that have colonized the substrates are identified and enumerated (Davies and Tsomides 1997). Aquatic life classification standards for a given water body are evaluated using numeric biological criteria that were statistically derived from the statewide database. The criteria are in the form of a statistical model (linear discriminant model) which yields the probability that a test sample belongs to one of the 3 water quality classes, or non-attainment of the lowest class (Davies et al. 1995). The model uses a set of metrics derived from the species composition and abundance enumerated from the substrates.

Geography

The Presumpscot River is the outlet of Sebago Lake. The river flows through the most densely-populated county in the State of Maine, crossing the towns of Gorham, Windham, Westbrook, Portland, and Falmouth. The Presumpscot then empties into Casco Bay at the Martins Point Bridge (Figure 6-1).

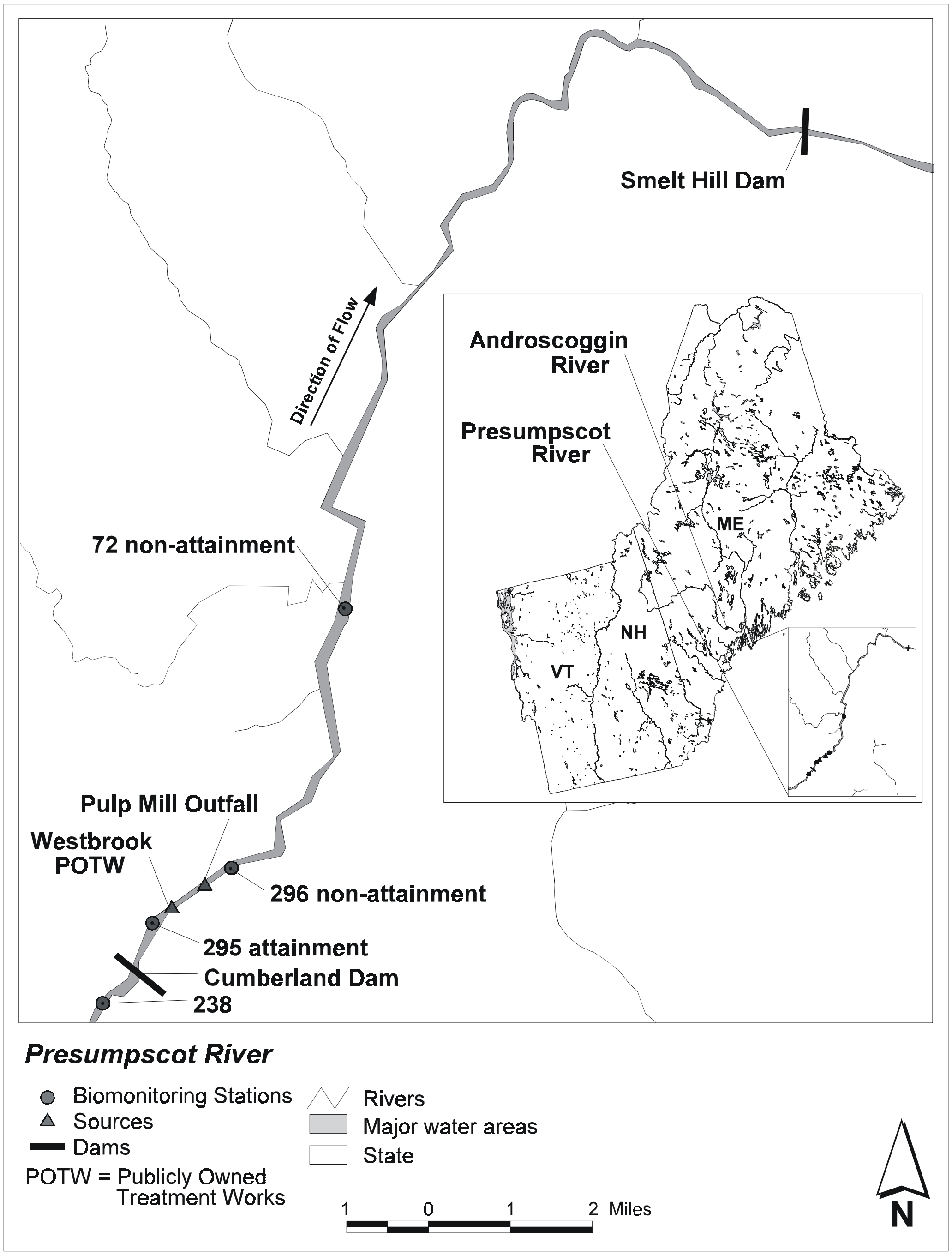


Figure 6-1. Map of the Presumpscot River showing biomonitoring stations, potential sources of impairment, and their location relative to the Androscoggin River (inset).

Compared to industrial receiving waters in the State of Maine, the Presumpscot is a relatively small river, having a drainage area of only 647 square miles. These circumstances contribute to a low dilution ratio in the lower Presumpscot River.

The river has six impoundments and four industrial and municipal waste discharges. This study comprises an area immediately downstream of a pulp and paper mill effluent discharge. Approximately 3.2 km, downstream of the discharge is an impoundment; upstream is a municipal discharge and (further upstream) two impoundments.

Evidence of Impairment

Biological Evidence: Biological monitoring in the Presumpscot River in Westbrook, below a pulp and paper mill discharge, consistently revealed non-attainment of Class C aquatic life standards (1984, 1994, 1995, 1996) using the standard Maine Department of Environmental Protection methods (Davies et al. 1995, Davies and Tsomides 1997).

Biological evidence indicating impairment on the lower Presumpscot River is summarized in Table 6-1 and Figure 6-2. Upstream samples consistently indicated attainment of Class C or better aquatic life standards (Davies et al. 1999). Three kilometers downstream the Presumpscot within the impounded area did not attain Class C aquatic life standards.

Table 6-1. Evidence of biological impairment in the Presumpscot River upstream and downstream of a pulp and paper mill effluent discharge.

Evidence	Upstream of Effluent	Downstream of Effluent
Aquatic Life Standard	Class C	Non-Attainment
Benthic Macroinvertebrate Community	90% insects	50% insects
Taxonomic Richness	--	15%-35% decrease relative to upstream
Sensitive Species (EPT)	--	46%-60% decrease relative to upstream
Snails and Worms	Low	High

The Presumpscot River biological monitoring samples reveal a shift in the benthic macroinvertebrate community from 90% insects above the mill to about 50% insects below the mill, with 15%-35% loss of taxonomic richness and 46%-60% loss of the sensitive Ephemeroptera-Plecoptera-Trichoptera (EPT) groups (Mitnik 1998). Pollution-sensitive insect taxa found in the upstream samples were replaced by a predominance of snails and worms, which are more tolerant of pollution, in the downstream samples.

6.3 List Candidate Causes

Eight candidate causes for the non-attainment of biological standards were considered. The candidate causes for the biological impairment of the Presumpscot River are shown in terms of a conceptual model (Figure 6-3), wherein the candidate causes are ordered from left to right. Each scenario is explained below:

1. Excess Toxic Chemicals - Potentially toxic compounds may be discharged from the paper mill and these compounds adversely affect aquatic life.

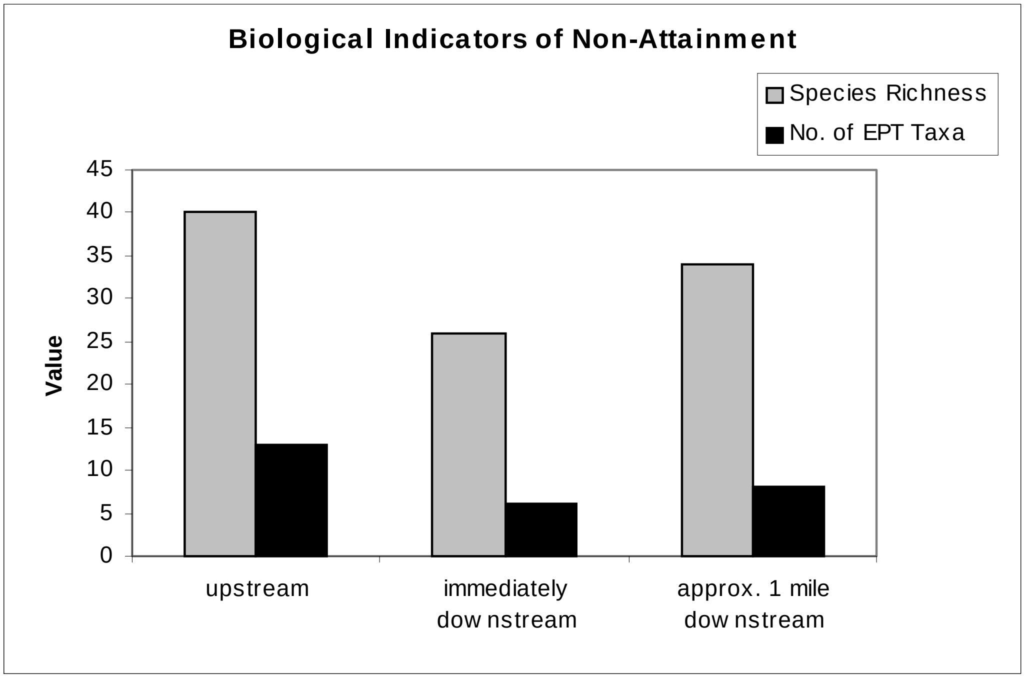
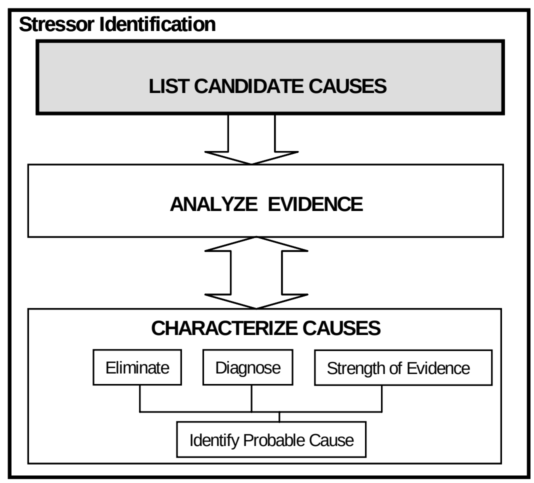


Figure 6-2. Species richness and number of EPT taxa in the Presumpscot River upstream and downstream of a pulp and paper mill effluent discharge.

2. BOD (produced by high TSS with floc) reduces DO - Excess total suspended solids (TSS with floc) may be released by the paper mill effluent, and these solids create biological oxygen demand (BOD), reducing dissolved oxygen (DO) levels in the river. Consequently, the river has insufficient oxygen to support sensitive species of benthic invertebrates.



3. TSS with floc - The increased levels of TSS discharged to the river could impact the benthic communities by accumulating as (non-biodegradable) sediment, resulting in fewer interstitial spaces in which animals can live, and possibly smothering benthic biota.

4. Excess Nutrients - Excess nutrients, deriving from either upstream, non-point sources or from the paper mill effluent, may affect water quality by promoting algal blooms. In this scenario, an overabundance of plant nutrients such as phosphorus is delivered to the stream, and over-stimulates algal growth (a process known as *eutrophication).* An increase of algae in the river may affect benthic macroinvertebrates in two ways. If the algal growth is severe, the resulting detritus becomes a source of BOD, reducing dissolved oxygen levels in the river. If the growth is modest, the algae may still affect the benthic macroinvertebrate community by providing an increased food supply for opportunistic invertebrates that use algae as a food source. Consequently, the community would shift in such a way that the opportunistic species would thrive and outcompete other, less opportunistic species.

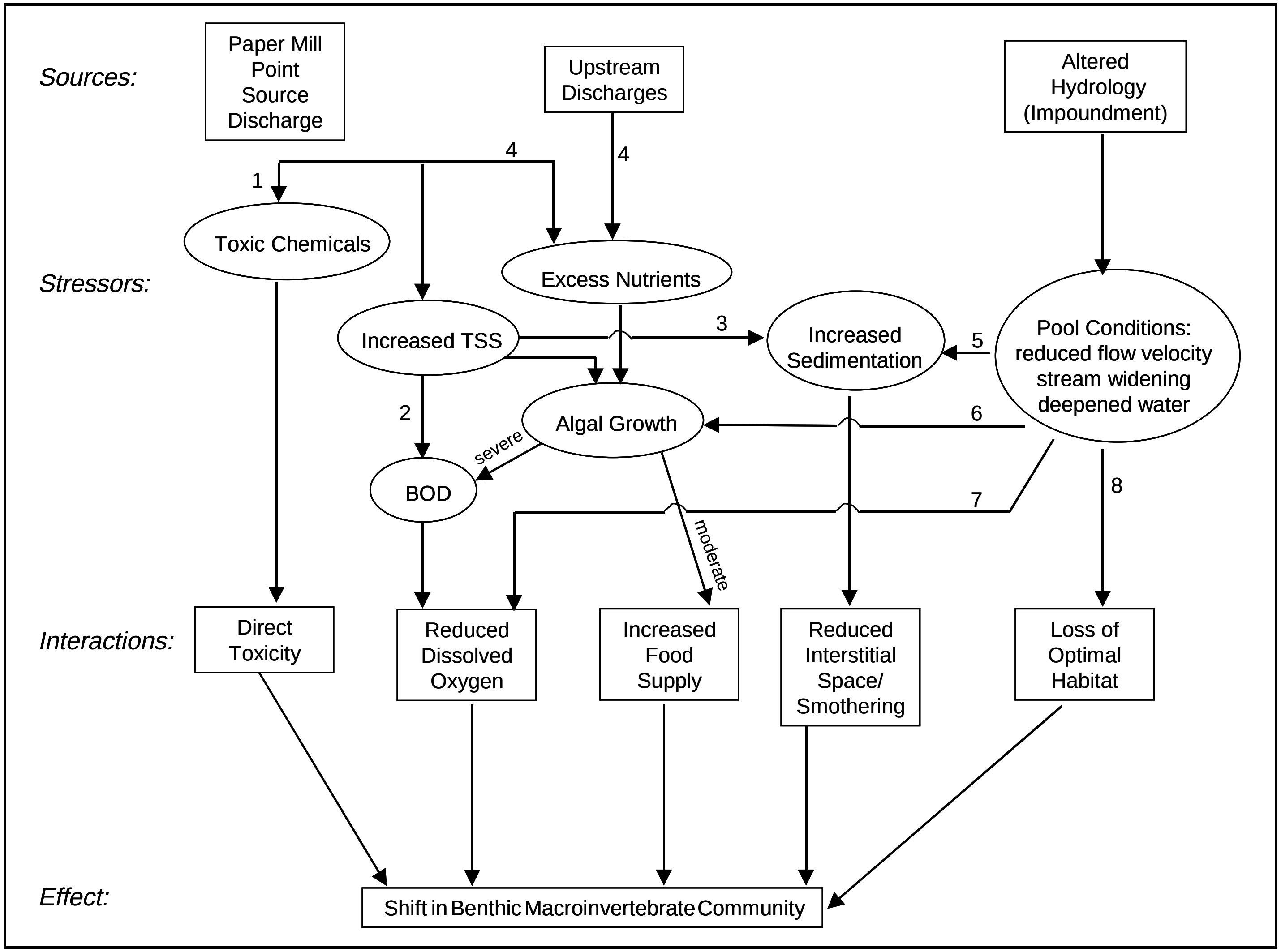


Figure 6-3. Conceptual model showing the potential impact of stressors on the benthic community of the Presumpscot River. (Arrow with minus sign (-) indicates inhibition.)

The fifth through eighth candidate causes are based on impoundment of the river just downstream of the paper mill effluent. Each cause begins with the idea that the impoundment is causing adverse changes in the physical nature of the Presumpscot. Impoundments generally widen and deepen a stream corridor, reducing flow velocity and creating pool-like conditions. Such alterations can have several effects:

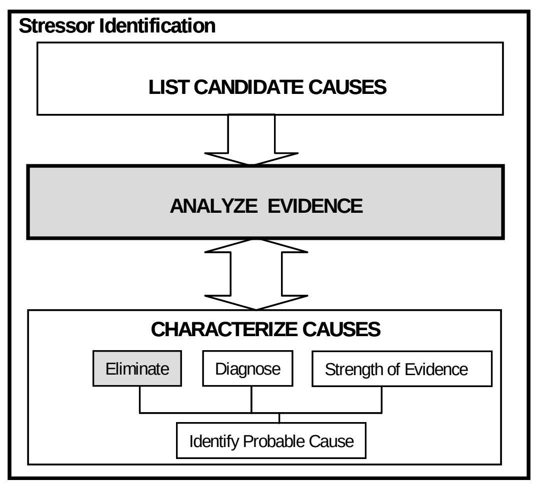
5. Impoundment Increases Sedimentation - One effect of impoundment is increased sedimentation due to reduced flow velocity, which leads to fewer interstitial spaces in which animals can live, and potentially smothers benthic ones.

6. Impoundment Promotes Algal Growth - The pool-like conditions created by the impoundment become a better habitat for algal growth, and algal blooms occur. Subsequently, benthic communities shift as a result of oxygen depletion or the dominance of algae-consuming invertebrates, as described previously.

7. DO Reduction in Impoundment - An impounded river is deeper and slower, which results in less potential for mixing and more potential for stratification, particularly in warmer months. As a result, underlying water may not be sufficiently aerated, and benthic diversity decreases in response to low dissolved oxygen levels.

8. Habitat Degradation caused by Impoundment - Changes in physical conditions of the river caused by impoundment reduce optimal habitat for benthic organisms. The effect is a direct one: native benthic macroinvertebrates are unable to thrive under the altered conditions. Dissolved oxygen levels and other water quality parameters are not a factor.

6.4 Analyze Evidence and Characterize Causes: Eliminate



Physical and Chemical Evidence: Physical and chemical evidence indicating impairment on the lower Presumpscot River is summarized in Table 6-2. Upstream of the pulp and paper mill outfall, it was possible to see samplers on the river bottom at 2.5 meters of depth, whereas in the effluent plume, just 600 m downstream, visibility was less than 0.5 meter. Visibility at a sampling station 3.2 km downstream of the outfall remained significantly impaired. This evidence was used to eliminate candidate causes.

1. Toxic Chemicals - No in-stream or sediment chemistry data were available. Therefore, toxic chemicals cannot be eliminated as a candidate cause.

Table 6-2. Physical and chemical parameters measured in the Presumpscot River upstream and downstream of a pulp and paper mill effluent discharge.

Observation	Source	Upstream of Mill	Downstream of Mill
Visibility	Mitnik 1998	2.5 m	<0.5 m (600 m below outfall) and visibility remained "significantly impaired" 3.2 km downstream
Observations on Sampling Equipment (e.g., ropes, nets)	Mitnik 1998	Free of brown floc	Coated with brown floc
Mean TSS (ppm)[1]	Courtemanch et al. 1997	3 ppm	5.9 ppm
Mean BOD (ppm)[2]	Mitnik 1994	3.96	6.19
DO range (ppm) [3]	Mitnik 1994	5.9 - 8.4	5.8 - 8.0
Mean nitrate - nitrite (ppm)[2]	Mitnik 1994	0.03	0.05
Mean ammonia (ppm)[2]	Mitnik 1994	0.03	0.12
Mean Total phosphorus (ppb)[2]	Mitnik 1994	12.8	61.2
Mean Orthophosphate (ppb)[2]	Mitnik 1994	3.5	44.3
Mean Chlorophyll *a* (ppb)[2]	Mitnik 1994	2.1	2.3

Notes
1 Observations from 1995-96; number unknown
2 4 sites above mill and 5 sites below on 3 consecutive days
3 Bottom water; 9 sites above mill and 8 sites below on 6 non-consecutive days

2. BOD (produced by high TSS with floc) reduces DO - Elevated BOD was associated with the biological impairment in the Presumpscot River. In this candidate scenario, reduced DO is the actual stressor that acts on the organisms to cause impairment. Monitoring in the Presumpscot River above and below the mill discharge indicated that DO concentrations did not decrease upstream to downstream (Table 6-2 and Figure 6-4). The reported DO measurements were taken at stations indicated on the map (Figure 1). Most of the sites shown in Figure 1 were impounded water; only 7.7, and 6.3 were free-flowing. The results reported in Tables 6-2 and 6-3 were all sampled between 0640 and 0850 hours, within 1m of the bottom, in July and August, 1993 (Mitnik 1994). This is the time, depth, and season at which minimum DO is found in lakes and impoundments, because of the diurnal cycle of photosynthesis and respiration, and because photosynthesis (but not respiration) is inhibited in deeper and darker waters. This analysis demonstrated that low DO does not occur in the Presumpscot River under any of the candidate causes involving reduced dissolved oxygen. **Therefore, candidate causal scenarios # 2** (High TSS with floc causes high BOD and reduced DO) **# 6** (Impoundment promotes algal growth that in turn reduces dissolved oxygen), **and # 7** (Impoundment causes low DO through decreased water flow rate) **could be eliminated without further analysis.** The elimination of scenario #6 is reinforced by the evidence described in scenario #4, below.

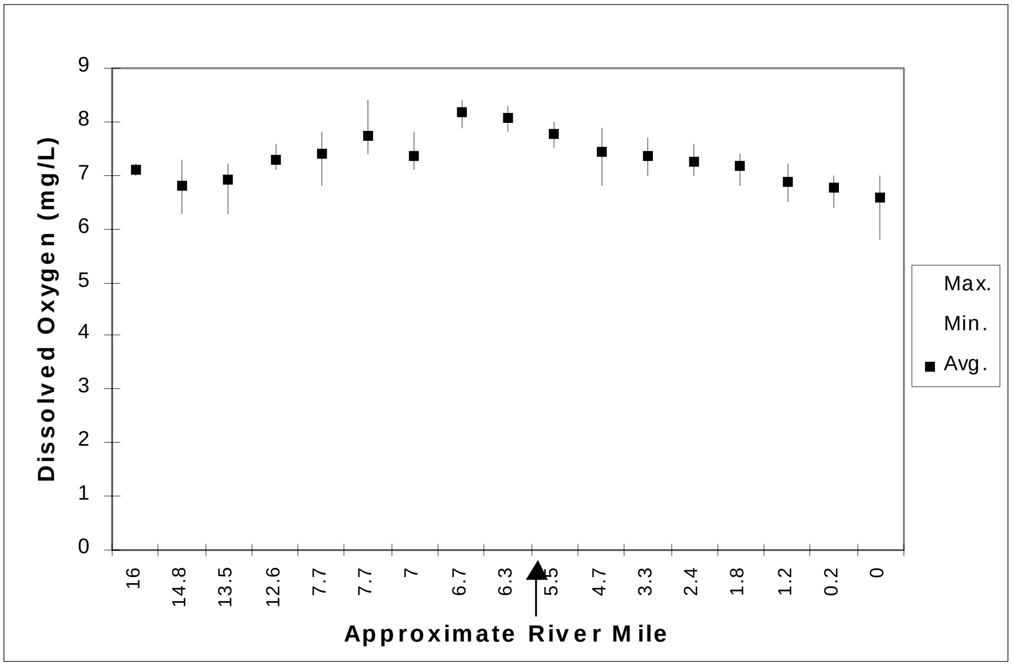


Figure 6-4. Bottom dissolved oxygen concentration in the Presumpscot River. (Means of 6 observations on 6 days in July and August, 1993, for each site. All observations within 1m of bottom. Whiskers extend to minimum and maximum measurements. All measurements were taken between 06:40 and 08:50 am. Arrow indicates location of the pulp and paper mill discharge: sites to the left of arrow are incrementally upstream of the discharge, and all sites to the right are incrementally downstream. The darkened square represents the average DO measurement. The lines above and below the square represent the maximum and minimum measurements, respectively.)

3. TSS with floc - In the Presumpscot River, TSS and floc are elevated at the impaired site (Tables 6-2 and 6-3). TSS with floc cannot be eliminated as a candidate cause.

4. Nutrients and Algal Growth - The nitrogen to phosphorus ratio ((TKN + NO_3 + NO_2)/TP) upstream of the paper mill discharge was approximately 25, indicating phosphorus limitation, as is typical of New England fresh waters. Below the paper mill discharge, elevated phosphorus concentrations were associated with biological impairment (Table 6-2). Specifically, total phosphorus (TP) and ortho-phosphate (PO_4) increase 5- to 10-fold downstream of the discharge (Table 6-2). Moreover, the discharge alone contained an average of 723 mg/L TP. A five-fold increase in TP (ten-fold in PO_4) would normally result in increased algal growth (measured as chlorophyll *a* concentration). However, the observed increase in chlorophyll *a* with the phosphorus enrichment below the discharge was negligible, increasing by just 0.2 ppb. Because excess algal growth is necessary for the causal pathway to be complete, **causal scenario #4 was eliminated from further consideration.**

5. Impoundment Increases Sedimentation - Biological impairment downstream of the paper mill discharge coincided with the presence of an impoundment (Table 6-3). However, no measurements of sediment loadings were available to determine if the biological impairment was the result of increased sedimentation caused by the impoundment. Therefore, scenario # 5 could not be eliminated.

8. Habitat Degradation caused by Impoundment - Again, the biological impairment found downstream of the paper mill discharge coincided with the presence of an impoundment (Table 6-3). However, no measurements of habitat quality were available to determine if the biological impairment was the result of habitat loss caused by the impoundment. Therefore, scenario #8 could not be eliminated.

Following the process of elimination, 4 candidate causes remained:

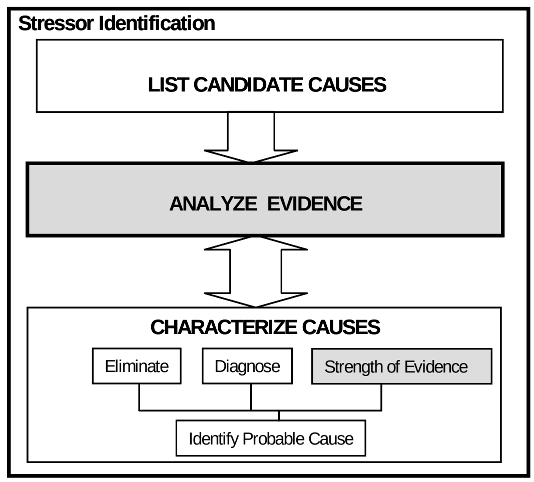
1. Excess toxic chemicals.

3. High TSS with floc causes smothering.

5. Impoundment increases sedimentation that smothers biota (with or without discharge of TSS and floc).

8. Impoundment causes loss of suitable habitat.

6.5 Analyze Evidence and Characterize Causes: Strength of Evidence



Direct observations in the Presumpscot River during macroinvertebrate and fish tissue sampling revealed a heavy suspended and settled solids load. Samplers and gill nets were coated with flocculent fibers and water clarity was dramatically reduced. In comparison to other paper mills in the State, the pulp and paper mill effluent released to the Presumpscot was considered high strength for solids. However, the conditions faced on the Presumpscot were similar to those found below the discharge from another paper mill on the Androscoggin River in Jay, Maine. Because of this, observations in the vicinity of the paper mill on the Androscoggin River were used to support the evidence found for this case study.

A comparison of the two rivers and discharge loadings to each is given in Table 6-4. Paper mill discharges on both rivers were subject to impoundments with similar hydraulic properties (e.g., velocity and depth) and background TSS concentrations (about 3 ppm). Two or more dams impounded both rivers upstream of the discharges.

Table 6-3. Considerations for eliminating candidate causes.

Candidate Cause	**Impairments occur same place as exposure?**	**Exposure increased over closest upstream location?**	**Gradient of recovery at reduced exposure?**	**Exposure pathway complete?**	**Candidate Causes Remaining**
Toxic Chemicals	NE	NE	NE	NE	X
BOD (produced by TSS) reduces DO	BOD Yes; TSS Yes; DO No	BOD Yes; TSS Yes; DO No	NE	No	
TSS with floc	Yes	Yes	NE	Yes	X
Nutrients and algal growth	Nutrients Yes; Algal Yes	Nutrients Yes; Algal No	NE	No	
Impoundment increases sediment	Yes	NE	NA	NE	X
Impoundment promotes algal growth	Algal Yes; DO No	Algal No; DO No	NA	No	
DO reduction in Impoundment	Imp. Yes; DO No	Imp. Yes; DO No	NA	No	
Habitat degradation caused by impoundment	Yes	NE	NA	NE	X

Table 6-4. Comparison of TSS loadings in the Presumpscot and Androscoggin Rivers. (Sample points were located below a pulp and paper mill effluent discharge.)

	Presumpscot		**Androscoggin**		
Mill & Year Sampled	1995	1996	1995	1996	1997
Aquatic Life Status	Non-Attainment	Non-Attainment	Non-Attainment	Attainment	Attainment
TSS treatment	none	none	none	TSS removal	TSS removal
Sampling Months	June-Aug	Aug-Sept	June-Aug	Aug-Sept	June-Aug
Flow, cubic feet/second (cfs)	418	463	2114	2982	4116
TSS Discharged, pounds/day	7454	8795	19804	5750	13495
TSS discharged/flow	3.31	3.52	1.74	0.36	0.61

Moreover, the upstream impoundments on both rivers attained at least Class C aquatic life standards. However, both rivers were found to be in non-attainment of aquatic life standards downstream of the paper mill discharges in 1995. Calculated mean ambient concentrations of TSS in the Presumpscot downstream of the mill were 32% to 39% greater than ambient levels downstream of the mill on the Androscoggin River. For the most part, the incremental TSS increase on the Androscoggin River, due to paper mill discharges, was within 1 ppm of background, while on the Presumpscot, the mill discharge was about 3 ppm greater than background.

In 1996, efforts were made by another paper mill on the Androscoggin River to reduce TSS discharge into the Androscoggin River. Following these efforts, the site's biological score improved and the river met Class C aquatic life standards. This recovery of biological conditions following TSS reduction provided experimental evidence that TSS could also be the cause of ecological stress in the Presumpscot River. Table 6-6 summarizes the types of evidence weighed in the analysis of potential stressors in the Presumpscot River.

Other evidence used in the strength of evidence comparison is shown in Table 6-5. Some metals exceeded chronic criteria when the maximum concentration in the effluent was evaluated with a low flow scenario (Table 6-5). Although low DO was eliminated in the previous step of this case study. Maine DEP performed an extensive modeling effort to investigate the potential for low DO below the mill outfall. The modeling results supported the conclusion that the DO concentrations did not fall below minimum levels for Class C aquatic life uses (Mitnik 1998). Furthermore, during the same time period as the biological monitoring, there were not violations of criteria for DO.

Table 6-5. 1996 - 1999 metal concentrations in the pulp and paper mill effluent.

Metals	Range µg/L in Effluent Grab Samples 1996-1999	Maximum Receiving Water Concentration (µg/L) at Low Flow[1] (7Q10[2])	Chronic Criteria (µg/L)	Acute Criteria (µg/L)
Aluminum	108 - 1920	207.9	87	750
Lead	3 - 14	1.52	0.41	10.52
Mercury	0.0001 - 0.9	0.097	0.012	2.4
Silver	10	1.083	0.12	0.92

Notes
1 The receiving water concentration is calculated from the maximum effluent concentration divided by a dilution factor of 9.
2 7Q10 + 7-day low flow over a ten year period.

Table 6-6: Strength of evidence of non-attainment in the Presumpscot River.

	Consideration	**TSS with Floc**		**Toxic Compound**		**Impoundment increases Sedimentation**		**Impoundment causes Loss of Habitat**	
		Results	**Score**	**Results**	**Score**		**Score**		**Score**
Case-Specific Evidence:	**Spatial Co-occurrence**	Compatible: Non-attainment observed in area of high TSS and floc loading. Attainment observed in upstream areas without TSS loading.	+	Evidence unavailable.	NE	Uncertain: Non-attainment observed in area of impoundment, but no measurements of sedimentation were available.	0	Uncertain: Non-attainment observed in area of impoundment, but no observations of habitat quality were available.	0
	Temporality	No observations prior to paper mill discharge.	NE	No observations prior to paper mill discharge.	NE	No observations prior to impounding	NE	No observations prior to impounding	NE
	Consistency of Association	No evidence	NE	No evidence	NE	A site within the same impoundment, upstream of the mill met aquatic life uses.	–	A site within the same impoundment, upstream of the mill met aquatic life uses.	–
	Biological Gradient	No evidence	NE	No evidence	NE	Not Applicable	NA	Not Applicable	NA
	Complete Exposure Pathway	Evidence for all steps: High TSS and floc discharge into river well-documented.	++	No evidence	NE	No evidence	NE	No evidence	NE
	Experiment	No evidence	NE	No evidence	NE	No evidence	NE	No evidence	NE

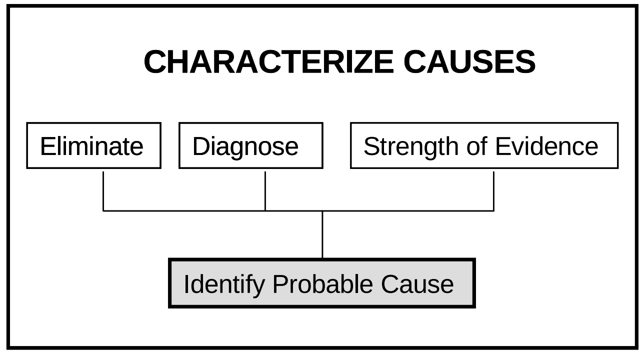
Table 6-6 (continued): Strength of evidence for causes of non-attainment in the Presumpscot River.

	Consideration	TSS with Floc		Toxic Compound		Impoundment increases Sedimentation		Impoundment causes Loss of Habitat	
		Results	**Score**	**Results**	**Score**		**Score**		**Score**
Information from Other Situations or Biological Knowledge:	**Plausibility - Mechanism**	Plausible: Snails and worms are adapted to utilization of settled solids.	+	Plausible: Toxic compounds could alter community composition.	+	Plausible: Sediment could alter habitat and community composition.	+	Plausible: Altered habitat could change community composition.	+
	Plausibility - Stressor-Response	TSS response from Androscoggin study and modeling sufficient to cause impairment.	++	Ambiguous: Assuming low flow conditions and at the highest concentrations reported for effluent from the mill, chronic aquatic life criteria might be exceeded for aluminum, lead, mercury and silver. However, if we assume high flows at the time of sampling then neither acute nor chronic aquatic life criteria are likely to be exceeded.	0	Other impoundments with similar potential sediment loadings support diverse invertebrate communities.	_	Other impoundments with similar habitat support diverse invertebrate communities.	_
	Consistency of Association	Invariant: Other sites on other rivers with TSS have impaired biological communities.	+++	In some places: Possibly could cause effects if at maximum values most of the time, but unlikely	0	Other impoundments on other rivers are not impaired.	_	Other impoundments on other rivers are not impaired.	_
	Specificity of Cause	Low: Other causes elicit similar responses.	0	Low: Other causes elicit similar responses.	0	Low: Other causes elicit similar responses.	0	Low: Other causes elicit similar responses.	0
	Analogy	No evidence	NE	No evidence	NE	No evidence	NE	No evidence	NE
	Experiment	Concordant: Removal of TSS in the Androscoggin river improved invertebrate assembleges.	+++	No evidence	NE	No evidence	NE	No evidence	NE

Table 6-6 (continued): Strength of evidence for causes of non-attainment in the Presumpscot River.

	Consideration	TSS		Toxic Compound		Impoundment increases Sedimentation		Impoundment causes Loss of Habitat	
		Results	**Score**	**Results**	**Score**		**Score**		**Score**
	Predictive Performance	No evidence	NE	No evidence	NE	No evidence	NE	No evidence	NE
Considerations Based on Multiple Lines of Evidence:	**Consistency of Evidence**	All Consistent.	**+++**	Not consistent: data collected during the same time period as the biological monitoring indicated that there were no violations of criteria for toxic materials (Mitnik 1998).	0	Not consistent: Other sites with impoundments maintained diverse communities.	0	Not consistent: Other sites with impoundments maintained diverse communities.	0
	Coherence of Evidence			Could be due to unmeasured chemical or episodic exposure.	0	No known explanation.	0	No known explanation.	0

6.6 Characterize Causes: Identify Probable Cause



Following the process of elimination, four causal scenarios remained to compare for strength of analysis (Table 6-6). These scenarios were: #1 (excess toxic chemicals), #3 (high TSS with floc causing smothering), #5 (impoundment increasing sedimentation that smothers biota, with or without discharge of TSS and floc), and #8 (impoundment causing loss of suitable habitat).

The evidence supporting scenario #3, that non-attainment was due to high TSS loads combined with floc, was consistent throughout the lines of evidence. Moreover, the strength of association, spatial co-occurrence, plausible stressor-response and experiment lines of evidence strongly supported this scenario. Therefore, high TSS with floc was sufficient for causing the biological impairment. The quality of the data are adequate for this conclusion, and our confidence is high.

In contrast, evidence for the toxicity scenario was weak, because the stressor-response association was unlikely based on levels of chemicals in the effluent and the likely dilution provided by the river at the time of discharge. If greater certainty was required, ambient receiving water toxicity tests could be used.

Likewise, evidence for the candidate causes involving impoundments lacked field measurements of sedimentation and habitat quality. However, our confidence in rejecting these scenarios as the primary cause of impairment is strengthened by the fact that several upstream sites along the Presumpscot River were impounded with no associated biological impairment (Mitnik 1998, Davies et al. 1999), and within the same impoundment upstream from the mill, the Presumpscot met aquatic life uses. Furthermore, several other impounded rivers of the state are able to meet Class B and C biological criteria (Davies et al. 1999).

Nutrient levels were elevated; however, the algal concentration was not different from the nearest upstream sampling location. As a result, candidate cause # 4, excess nutrients, was eliminated; however, it is possible that the growth of algae was inhibited by other factors, such as shading from floc. If floc were removed, then effects due to eutrophication might become evident.

Low dissolved oxygen was also eliminated based on spatial patterns of DO along the river. Other data is also available that increases the confidence that could have been presented in a strength of evidence analysis. At the site, DO was not below 6 ppm. The minimum DO level for Class C waters is 5 ppm. Maine DEP also performed an extensive modeling effort to investigate the potential for low DO below the mill outfall. The modeling results supported the conclusion that the DO concentrations did not fall below minimum levels for Class C aquatic life uses (Mitnik 1998).

6.7 Significance and Use of Results

In December 1998, the U.S. Environmental Protection Agency approved a Total Maximum Daily Load (TMDL) finding, prepared by Maine Department of Environmental Protection, for the Presumpscot River. This approval was significant for several reasons:

1. It was the first TMDL that addressed a listed 303(d) water to be approved in Region 1 USEPA (the New England States);

2. It was the first time in New England that bioassessment findings had served as the quantitative response variable from which a pollutant discharge limit was developed.

The wastewater discharge license that has resulted from this effort requires an initial 30% reduction in the TSS discharge from a pulp and paper mill in Westbrook. Provisions are included in the license for further reductions (up to 61%) if the initial levels still fail to provide for attainment of aquatic life standards.

Main Factors Influencing Success

The Department was able to apply this innovative approach to improving water quality and aquatic life conditions in the Presumpscot River because of the convergence of several factors:

- The State has a sound legal basis for use of biological monitoring findings to force action. Clearly defined aquatic life standards exist in the Water Quality Classification law and technically-defensible numeric criteria have been established by the Department;

- Data essential to the modeling of the recommended total suspended solids load reductions on the Presumpscot River had been collected to assess aquatic life issues on the Androscoggin River (under State requirements for a 401 Water Quality certification for a hydropower license renewal);

- Teamwork and collaboration between DEP, water quality modelers, and aquatic biologists resulted in an approach that integrated technical information and expertise from both disciplines. It also provided a means for the Department to control a stressor (TSS) for which the State has no standards.

6.8 References

Courtemanch, D.L., P. Mitnik, and L. Tsomides. 1997. Dec. 8, Memorandum to Greg Wood, Maine Department of Environmental Protection Licensing Section, Augusta, Maine.

Davies, S.P., L. Tsomides, D.L. Courtemanch, and F. Drummond. 1995. *Maine biological monitoring and biocriteria development program.* Maine Department of Environmental Protection, Augusta, Maine.

Davies, S.P. and L. Tsomides. 1997. *Methods for biological sampling and analysis of Maine's inland waters.* DEP-LW/07-A97. Maine Department of Environmental Protection, Augusta, Maine.

Davies, S.P., L. Tsomides, J.L. DiFranco, and D.L. Courtemanch. 1999. *Biomonitoring retrospective: Fifteen year summary for Maine rivers and streams.* DEPLW1999-26. Maine Department of Environmental Protection, Augusta, Maine.

Hilsenhoff, W.L. 1987. An improved biotic index of organic stream pollution. *Great Lakes Entomol.* 20:31-39.

Mitnik, P. 1994. *Presumpscot River waste load allocation.* Maine Department of Environmental Protection, Augusta, Maine.

_____. 1998. *Presumpscot River supplemental report to waste load allocation.* Maine Department of Environmental Protection, Augusta, Maine.

Chapter 7

Little Scioto River, Ohio

7.1 Executive Summary

This case study of the Little Scioto River represents an application of the SI process to a complicated system. Impairment of the Little Scioto River reflected several impacts caused by different stressors. Originally, the data on the Little Scioto were collected and analyzed as part of the Ohio Environmental Protection Agency (OEPA) state monitoring program during 1987, 1991, 1992 and 1998 (OEPA 1988b, 1992, 1994, unpublished data from 1998) and as research for a USEPA methods development program. The monitoring data were subsequently analyzed for this SI case study to demonstrate how data collected from monitoring programs could be used to identify probable causes of biological impairment.

The SI investigation was initiated because criteria in the state of Ohio's water quality standards were violated in parts of the Little Scioto, a small river in north-central Ohio (Yoder and Rankin 1995b). The SI investigation involved a 9-mile stretch of the Little Scioto River near Marion, Ohio, where there was evidence of biological impairment.

The State of Ohio has a "tiered" set of aquatic life use designations based on narrative definitions of specific aquatic uses that are protected by a set of numeric biocriteria, chemical criteria, and habitat criteria. Ohio EPA determines biological impairment of stream segments by comparing study sites to the numeric biocriteria in their water quality standards. OEPA uses standard multimetric indices, including the Index of Biotic Integrity (IBI), the Invertebrate Community Index (ICI) (OEPA 1989a), and the Qualitative Habitat Evaluation Index (QHEI) (OEPA 1989c). Little Scioto River data collected in 1987 and 1992 showed a condition of "fair" to "severe impairment" in the stretch from river mile (RM) 9.2 to where the Little Scioto joins the Scioto River, just downstream of RM 0.4.

<u>Describe the Impairment</u>

Three distinctive impairments (A, B, and C) were identified for the causal evaluation (at RM 7.9, 6.5, and 5.7, respectively). Impairment A was characterized by a loss of fish and benthic invertebrate species, a decrease in the number of individual fish, and an increase in the relative weight of fish. Impairment B was characterized by a decrease in the relative weight of fish and a large increase in deformities, fin erosion, lesions, tumors and anomalies (DELTA). Impairment C was characterized as having a further increase in DELTA and extirpation of a Tribe of midges, the Tanytarsini.

<u>List Candidate Causes</u>

Stressors impacting the upper portion of the river were identified as mostly non-point nutrient and sediment loadings associated with agriculture. Beginning at river mile 9.0 and continuing to the mouth, the river is channelized. The Little Scioto River at and below Marion, Ohio, however, has been notably contaminated with elevated levels of polycyclic aromatic hydrocarbons (PAH). Creosote and metals in sediment samples and

ammonia, phosphorous (P), total nitrogen (N) were detected in water samples (OEPA 1994).

Based on the knowledge about the site and effects, six candidate causes were hypothesized to account for the three major biological impairments observed in the Little Scioto study area:

1. Habitat alteration: embedded stream and deepened channel

2. Exposure to PAHs

3. Metal contamination

4. Ammonia Toxicity

5. Low Dissolved Oxygen/High Biological Oxygen Demand

6. Nutrient Enrichment

<u>Characterize Causes: Eliminate</u>

Candidate causes were eliminated because the level of exposure to the candidate cause did not increase compared to the nearest upstream location. Candidate causes that remained after the elimination step are listed below:

- Impairment A (RM 7.9) — habitat alteration, metal contamination, and nutrient enrichment remained as probable causes.

- Impairment B (RM 6.5) — PAH contamination, metal contamination, ammonia toxicity, low dissolved oxygen/high biological oxygen demand, and nutrient enrichment remained as probable causes.

- Impairment C (RM 5.7) — metal contamination, ammonia, and nutrient enrichment remained as probable causes.

<u>Characterize Causes: Diagnose</u>

No evidence strong enough to support diagnosis was available for any of the candidate stressors.

<u>Characterize Causes: Strength of Evidence</u>

A strength of evidence approach was used to examine the remaining causes with regard to each impairment. Evidence based on other situations and biological knowledge were especially important including consistency of association and plausibility of mechanism and stressor-response.

<u>Characterize Causes: Identify Probable Causes</u>

<u>Impairment A</u>
At Impairment A, the increased relative weight is probably caused by the artificial deepening of the channel that allows larger fish to live there. The mechanisms were

probable, and consistency of association and experiments from other sites in Ohio and elsewhere supported this finding for the specific impairments. The extirpation of fish and benthic invertebrates seems to be most likely due to embedded substrates. Although low DO could also be a cause, upstream locations had even lower DO levels and yet had a greater variety of fish and invertebrate species.

Although metals were present, the likelihood of response at these concentrations is low. Furthermore, the types of changes in the community, especially an increase in the relative weight of fish, is very unlikely with the candidate cause of metals. Although P levels are slightly higher, effects are not associated with these phosphorous concentrations elsewhere, and they do not exceed Ohio proposed criteria values for effects. PAH and ammonia had already been eliminated because levels were the same or lower than upstream. Low DO /BOD was also eliminated as an overall pathway; however, low DO associated with channelization may still play a role, especially with respect to the slight increase in the percentage of DELTA.

<u>Impairment B</u>
A single probable cause, toxic levels of PAH-contaminated sediments, is likely for the three manifestations of Impairment B: decreased relative weight, increased DELTA, and decreased species. All of the evidence support PAH contamination as the cause. There is a complete exposure pathway at the location, and a clear mechanism of action for each of the effects. The single most convincing piece of evidence is that the cumulative toxic units of PAH were more than 300 times the probable effects level.

Metals are at sufficient concentrations to cause effects; however, they are at levels close to upstream concentrations, and are less than 2% as toxic as the lowest cumulative toxic units of PAH. Metal concentrations are high enough that they should be considered a potentially masked cause. Reduced DO resulting from increased BOD is unlikely because, downstream, even greater levels of BOD did not cause reduction of dissolved oxygen. Ammonia and nutrient enrichment are unlikely given that state criteria levels were met and given the much stronger evidence for PAH. Habitat alteration continues to impair the site, but it is not the cause of the increased DELTA, decreased relative weight, or the additional decline in the number of species, because the level of embeddedness was similar to upstream.

<u>Impairment C</u>
At Impairment C, increased % DELTA and % Tanytarsini may have different causes. Increased DELTA in fish is probably caused by increased P and N. Nutrients, especially P, have been associated with increased fin erosion and lesions, but some uncertainty exists since P acts indirectly. Another candidate cause is also probable, namely, ammonia. Ammonia is slightly higher at Impairment C than at Impairment B, and exceeded ammonia criterion values. Biological gradients were absent for ammonia; however, this may have been a statistical artifact given the number of sites available to perform the analysis, and the potential interference from other stressors downstream.

Metals are considered unlikely, because very specific surface lesions are only occasionally noted as effects from long-term exposure, and only some metal concentrations were slightly greater than at Impairment B. Metal concentrations are high enough that they should be considered a potentially masked cause.

The probable cause of extirpation of Tanytarsini at Impairment C is more uncertain because less is known about the natural history and stressor-response relationships of these benthic invertebrates. Nutrient enrichment still seems to be the most likely cause since all of the strength of evidence considerations were consistent.

PAH contamination and habitat alteration continue to impair the site, but they are not the cause of the increased percent DELTA or extirpation of Tanytarsini.

Identify Probable Cause

The most probable causes were:

- Impairment A (RM 7.9) — Siltation and deepened channel are consistent with impairment A. The magnitude of the alteration and clear difference from upstream locations strongly support this cause.

- Impairment B (RM 6.5) — PAH-contaminated sediments are likely causes for the three manifestations of Impairment B.

- Impairment C (RM 5.7) — The causal characterization at Impairment C is less certain, but the strength of evidence favors increased nutrient enrichment as the cause.

The Little Scioto case study is a good example of a complex system requiring a detailed analysis. Although it was possible to identify the dominant causes of specific impairments, other causes were present that had the potential to cause impairments if the dominant cause was removed. For instance, habitat alteration associated with channelization would still impair the entire river below RM 9.0.

7.2 Introduction

The Little Scioto case study involves a nine-mile stretch of a river suffering from several impairments with different causes. Typical of similar stressor investigations, the data examined for this case study were not collected or originally analyzed specifically for the Stressor Identification Technical Guidance Document. Rather, they were collected as a part of the Ohio EPA state monitoring program during 1987, 1991, 1992 and 1998 (OEPA 1988b, 1992, 1994, unpublished data from 1998), and as research for a USEPA methods development program. These monitoring data were subsequently analyzed in this study to demonstrate how data collected from existing monitoring programs could be used to identify probable causes of biological impairment.

Various types of data were used in this case study, including chemical analyses (sediment, water, and fish tissue) and biological assessment (biological community and physical habitat). Methods for the collection and analysis of chemical data are described in Ohio EPA (1989c). In 1992, one grab sample was taken, whereas in 1987, multiple grab samples were taken. Other Ohio EPA data sets included biological assessment data on fish and invertebrate assemblages and physical habitat measurements. In Ohio, impairment of stream aquatic life uses are defined by standard multimetric indices including the Index of Biotic Integrity (IBI) and the Invertebrate Community Index (ICI) (OEPA 1989a). These indices have been promulgated as numeric biocriteria in the State's water quality standards. The quality of the habitat is characterized using the Qualitative Habitat Evaluation Index (QHEI) (OEPA 1989c). These methods are described in detail by Ohio EPA (1989c). Biochemical measurements of impairment included bile metabolites measured according to Lin et al. (1996) and ethoxy resorufin[O]deethylase (EROD) activity measured according to Cormier et al. (2000b). Although the attempt was made to use biological and chemical data from the same locations, in some cases, chemical measurements were recorded at a location that did not exactly coincide with the location of biological assessment (e.g., RM 5.8 and RM 5.7, respectively). However, the distance between the chemical and biological sample sites was negligible or overlapped, and the data were able to be used to analyze associations between candidate causes and the biological impairment.

The Little Scioto River is a small river in north-central Ohio that empties into the Scioto River (Figure 7-1). It drains primarily farmland in the northeastern quadrant of the Eastern Corn Belt Plains ecoregion. The soils in this area are glacial till overlying limestone, dolomite, and shale bedrock. The water table has been lowered in much of the watershed by extensive use of tile drainage in crop fields. Near Marion, Ohio, the Little Scioto is biologically impaired.

This causal investigation was initiated because the State of Ohio water quality standards related to biological criteria were violated (Yoder and Rankin 1995a). The State of Ohio has a "tiered" set of aquatic life use designations based on narrative definitions of specific aquatic uses, which are protected by numeric criteria.

The majority of Ohio rivers and streams are designated as *Warmwater Habitat* (WWH) (Yoder and Rankin 1995a). This designation is narratively defined as *supporting a balanced, reproducing aquatic community*. Quantitatively, the minimum criteria required to be in attainment of WWH standards are defined as the 25th percentile values of reference condition scores for a given index, site type, and ecoregion. The choice of the 25th percentile is considered to be conservative and will likely be influenced by the inclusion of marginal sites as well as reference quality sites.

The Little Scioto River is considered Warmwater Habitat above RM 7.9 and a Modified Warmwater Habitat at and below RM 7.9 (see Figure 7-1). The *Modified Warmwater Habitat* (MWH) criteria are based on comparisons to a different reference condition than are used for the WWH criteria (Yoder and Rankin 1995a). The MWH designation is a *non-fishable* aquatic life use, and is designed to protect streams that have been too impacted, or modified, to meet WWH standards. MWH streams are unlikely to recover sufficiently to meet WWH designation. Consequently, MWH criteria are typically lower than WWH criteria. In spite of poorer water quality conditions (such as low dissolved oxygen, high ammonia concentration, and increased nutrient input), MWH streams are nonetheless able to support permanent assemblages of tolerant species.

7.3 Evidence of Impairment

In 1987 and 1992, sampling and measurements for community and habitat indices (IBI, ICI, QHEI) were conducted by OEPA along the Little Scioto River. Standardized field, laboratory and data processing methods followed OEPA procedural guidelines (OEPA 1988a, OEPA 1989a,b,c, Rankin 1989). Fish and macroinvertebrates were sampled at seven sites along the river, from river mile (RM) 9.5 to 0.4 (Figure 7-1). Index and metric scores for IBI, ICI, and QHEI used in this study were obtained from data sets that were generated and made available by OEPA as well as various OEPA reports (1988b, 1992, and 1994).

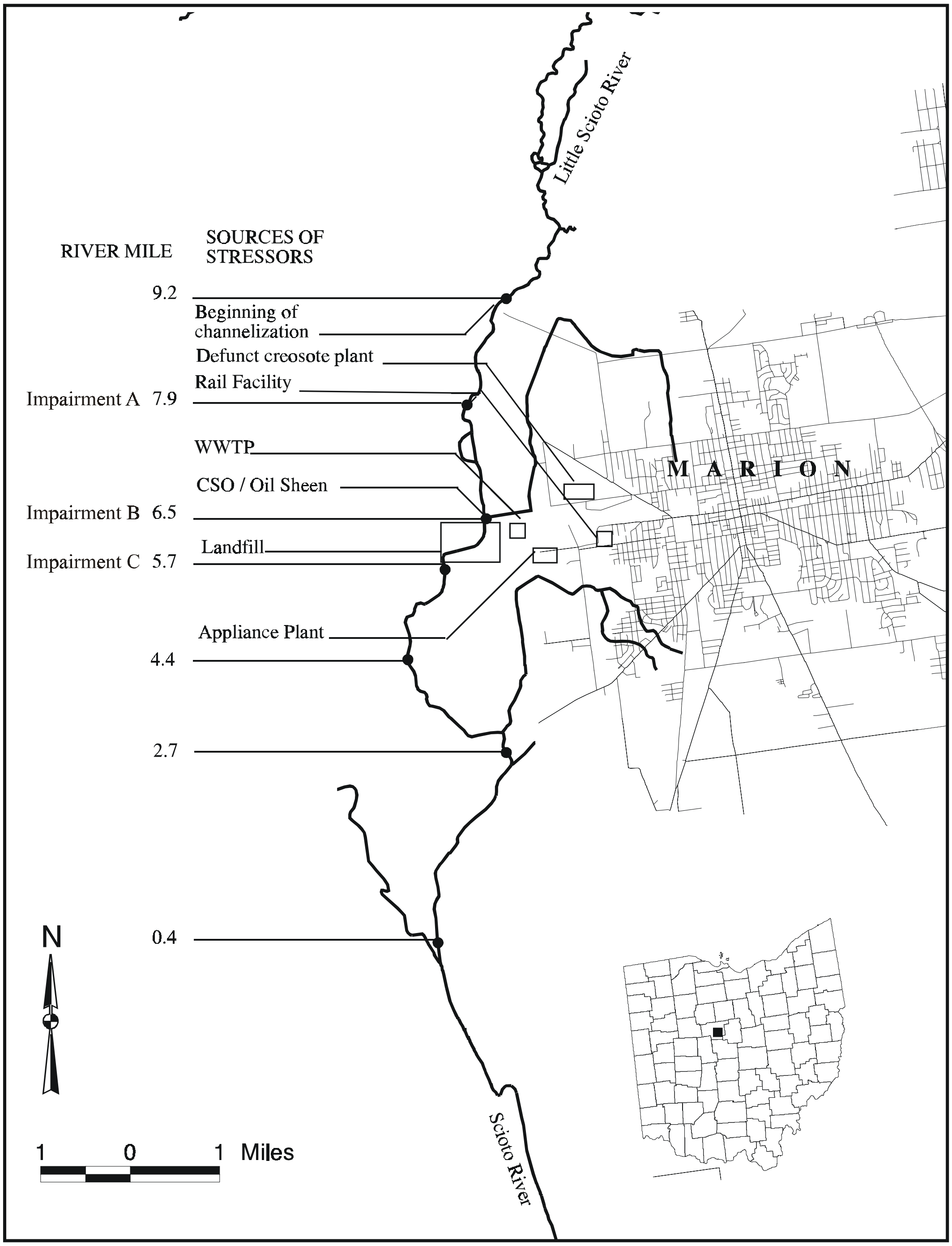


Figure 7-1. Map of the Little Scioto River, Ohio, showing sites where fish were sampled. (Approximate locations of significant physical features, tributaries and point source inputs are noted. The small inset shows the location of the study area in the state of Ohio. Locations of Impairments A, B and C are also shown.)

Of the seven sites sampled in 1987, the highest IBI score was 34 (out of a possible score of 60), which occurred at RM 9.2. This score translates to a *fair* ranking according to WWH standards. The remainder of sites were described as *severely impaired*, with IBI scores between 25 and 12 (the lowest possible IBI score) (OEPA 1994, Yoder and Rankin 1995a). In 1992, the IBI score at RM 9.2 decreased by one point to 33. However, in 1992, the IBI score dropped 9 points to a score of 24 between RM 9.2 and

RM 7.9. Another 5 point drop occurred at RM 6.5 and scores stayed between 19 and 20 through RM 2.7. At RM 0.4, the IBI score climbed back to 25, greater than the adjacent upstream site's score, but still indicating impairment. Figure 7-2A illustrates the fluctuation of the IBI at the seven sites during the two sampling years (1987 and 1992).

Figure 7-2B traces a similar pattern of impairment for the invertebrate index during the 1987 and 1992 sampling years. The ICI met WWH aquatic life use standards in 1987 and 1992 at RM 9.2, with scores of 40 and 38, respectively (Figure 7-2). In 1992, the ICI score declined 22 points at RM 7.9 with a score of 16, considered fair, but below MWH aquatic life use standards. Scores further declined 12 or more points at RM 6.5, 5.7 and 4.4, with scores ranging between 6 and 10. These scores are indicative of highly impaired conditions (OEPA 1994, Yoder and Rankin 1995a). ICI scores increased to a value of 18 downstream at RM 2.7 and RM 0.4. In 1987, both IBI and ICI scores were greater at RM 6.5 and then declined at RM 5.7, and remained very low to the mouth of the Little Scioto.

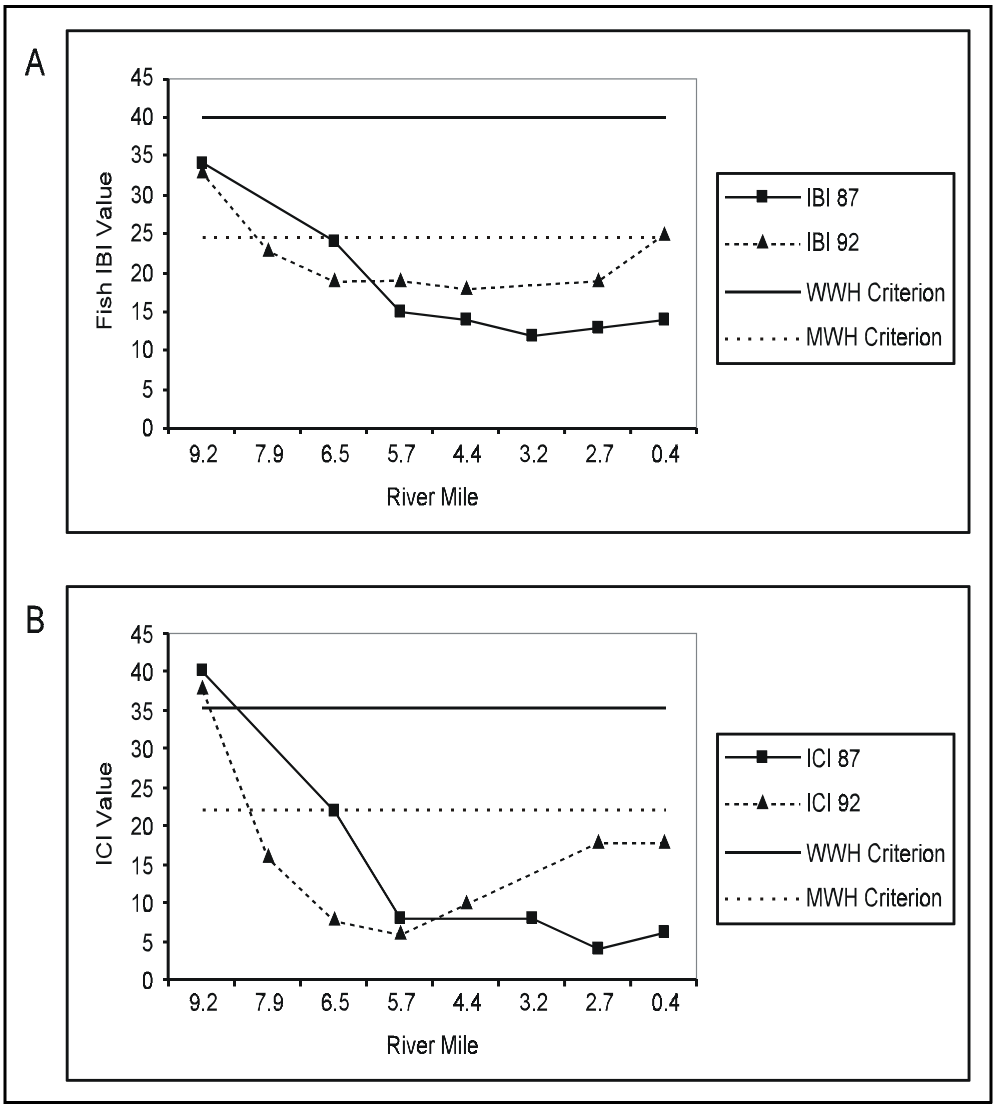


Figure 7-2. Spatial changes in fish IBI (A) and benthic macroinvertebrate ICI (B) values in the Little Scioto River in 1987 (OEPA 1988) and 1992 (OEPA 1994).

The impairments seen below RM 9.2 were more specifically described by examining the metrics that make up the IBI and the ICI. This information was combined with the changes seen in the overall IBI and ICI scores to determine whether distinctive patterns of impairment could be identified. Each distinctive impairment required a separate causal evaluation.

A subset of the fish and macroinvertebrate metrics, selected to highlight differences in community patterns, is shown in Figures 7-3A (fish) and 7-3B (macroinvertebrates). The complete list of values for the metrics is shown in Tables 7-13 and Table 7-14 (Please note that Tables 7-13 through 7-20 are located in Section 7.13, "Additional Data Tables"). One of the metrics, relative weight of fish, is not a component of the IBI but a component of another index, the Modified Index of Well-being (MIWB). Examination of the spatial distribution of the IBI, ICI, and metric patterns in 1992 indicates that at least three distinct impairments occurred:

- Impairment A was seen at RM 7.9 where a marked drop in both the IBI and ICI occurred relative to the upstream location at RM 9.2. Specific fish metrics that appeared to correspond to this drop included decreases in the number of individuals minus tolerant fish, decreased total number of species, and increased relative weight. In addition, the percentage of mayfly species decreased.

- Impairment B occurred at RM 6.5 and corresponded with an additional decrease in both the IBI and the ICI. Relative the upstream location at RM 7.9, fish relative weight decreased, the number of deformities, erosions, lesions, tumors and anomalies (DELTA) increased, and the percentages of mayflies and Tanytarsini midges also decreased while the percentage of tolerant organisms increased.

- Impairment C occurred at RM 5.7. There was no change in the IBI relative to RM 6.5, although relative weight of fish decreased and DELTA increased. The invertebrates had variable changes depending on the sampling year. In 1987 and 1992, the % Tanytarsini midges decreased or disappeared entirely. Changes in the metrics at these three locations are summarized in Table 7-1.

The biological assessment data for the remaining locations showed a pattern similar to Impairment C, with the possibility of intensification at RM 4.4 and some improvement in metric scores occurring at RM 2.7 and 0.4. A fourth impairment was not hypothesized for RM 4.4 because the pattern of fish and invertebrate metrics were fairly similar to those seen at RM 5.7.

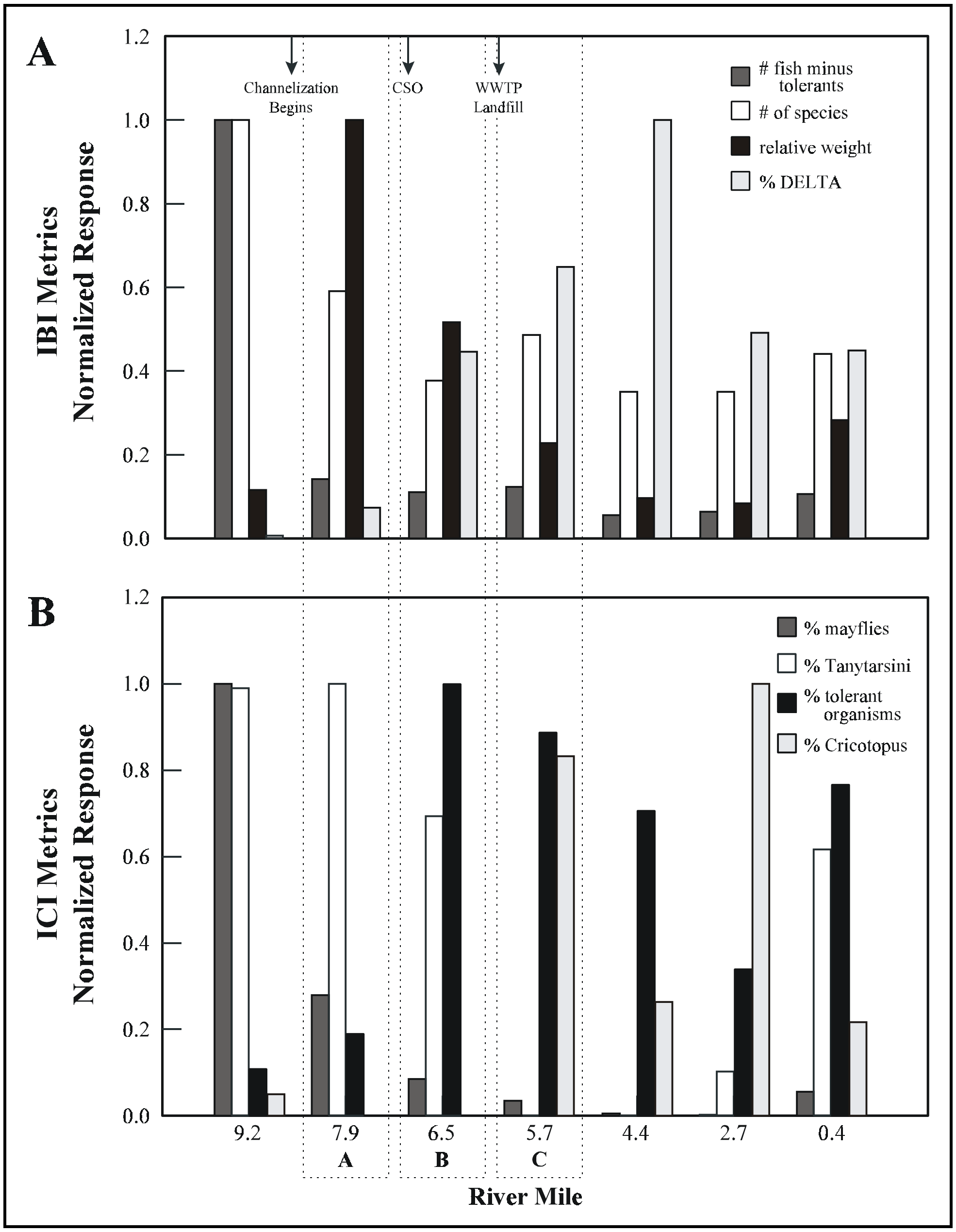


Figure 7-3. Changes in the IBI and ICI scores over distance in the Little Scioto River, 1992. ((A) Changes in the relative scores for the total number of individual fish minus tolerant fish (# fish minus tolerant), the number of species (# species), the relative weight of fish (relative weight) and the percentage of DELTA. (B) Changes in the relative abundances of percent *Ephemeroptera, Tanytarsini*, tolerant organisms, and *Cricotopus*, in the Little Scioto River. Normalized values were calculated by dividing the value at the individual site by the highest value for all sites.)

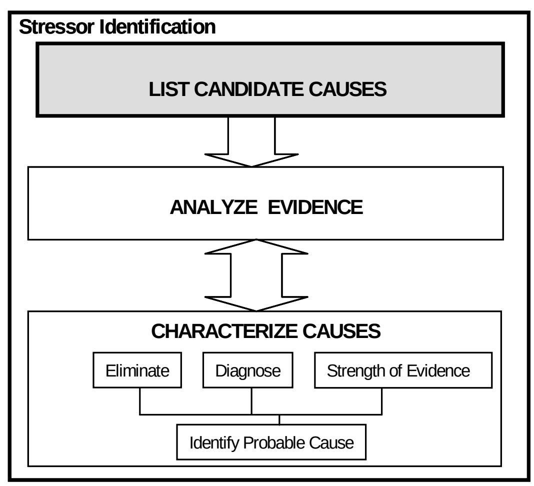
Table 7-1. Summary of the three impairments that were considered in the Little Scioto River. (Each location is scored relative to the location immediately upstream, based on 1992 data.)

Response	Impairment A RM 7.9	Impairment B RM 6.5	Impairment C RM 5.7
Fish			
# of individuals minus tolerant individuals	-	+	-
# Species	-	-	0
Relative Weight	7.9	-	-
DELTA	0	0	5.7
Invertebrates			
% Mayflies	-	-	-
% Tanytarsini midges	0	-	-
% Tolerant taxa	0	0	-
% *Cricotopus* sp.	-	0	0

(+) indicates an increase in the metric relative to the next upstream location
(-) indicates a decrease
(0) indicates no change.

7.4 List Candidate Causes

Evidence Used to Develop Candidate Causes



Many point and non-point sources of pollutants are associated with the Little Scioto River. Stressors impacting the upper portion of the river are mostly non-point nutrient and sediment loadings associated with agriculture. However, the Little Scioto River, at and below Marion, Ohio, has been notably contaminated with elevated levels of polycyclic aromatic hydrocarbons (PAH). Creosote and metals were found in sediment samples, and ammonia was detected in water samples (OEPA 1994). The OEPA has, in fact, recently requested Superfund support in the clean-up of an abandoned wood creosote plant suspected of polluting the river since the 1860's (Edwards and Riepenhoff 1998). An oily sheen was noted on the river between river miles 6.5 and 5.8 during a site visit in 1992 (Cormier, pers. observ.). In-stream habitat quality was also degraded by channelization that took place in the early 1900's (OEPA 1994). Locations of the potential sources and stressors, including a landfill and wastewater treatment plant (WWTP), are shown in Figure 7-1.

<u>List of Candidate Causes and Scenarios</u>

As noted previously, three distinctive impairments were identified for the causal evaluation. Based on the knowledge of the sources and effects, six candidate causes were formulated to account for the impairment observed at each site. A conceptual model of these candidates is provided in Figure 7-4.

1. Habitat Alteration - Habitat alteration, resulting from channelization, combines a complex interaction of several stressors. These stressors are evident at RM 7.9 and continue to the mouth of the river. Channelization can alter biological communities by changing the physical structure of the stream and the flow characteristics of the water, ultimately lowering dissolved oxygen, increasing siltation, and reducing substrate complexity. This complex suite of stressors also includes: decreased woody debris, which reduces available substrate and changes the energy source; decreased sinuosity, which changes flow characteristics; erosional patterns and substrates; increased channel depth that favors larger species of fish; loss of pools that act as refugia; and loss of riffles that oxygenate water and transport sediment (Tarplee et al. 1971, Karr and Schlosser 1977, Yount and Niemi 1990, Allan 1995).

2. PAH and 3. Metals - Biological impairment could also have been caused by toxic stress. Historically, the river has provided a means of waste disposal for various industries, whose effluents have contained metals, PAH, and creosote. Waste materials may have also been buried in the landfill below RM 6.5 (OEPA 1994). All are potentially toxic to aquatic life, and some have the ability to bioaccumulate through the food web (Eisler 2000a,b). Thus, two candidate causes emerge: candidate cause #2 is that biological impairment has occurred due to PAH exposure (with PAH emanating from creosote deposits), and candidate cause #3 attributes impairment to metal contamination.

4. Ammonia Toxicity - Ammonia is directly discharged into streams by point sources (Russo 1985, Miltner and Rankin 1998). Ammonia can also be formed as the result of nutrient enrichment. When dissolved oxygen levels are low, nitrates are reduced to ammonium ion. If pH is high, some of the ammonium ion is converted to un-ionized ammonia, which is toxic to aquatic organisms (Russo 1985). Moreover, pH may rise during periods of high photosynthetic rates from bicarbonate depletion. High amounts of nutrients often lead to increased algal growth rates, and the conversion of ammonium to un-ionized ammonia is expedited (Dodds and Welsh 2000).

5. Low Dissolved Oxygen/ High Biological Oxygen Demand - Depletion of DO commonly occurs from organic enrichment (Smith et al. 1999). Organic enrichment is the most common cause of increased biological oxygen demand (BOD) (Allan 1995). Potential sources of excess organic matter within the study area include a waste water treatment plant (WWTP) and several combined sewer outfalls (CSOs), as well as upstream, non-point sources. Organic matter is also produced by excess algal growth from nutrient enrichment (Dodds and Welsh 2000). Algal blooms themselves result in increased organic matter regardless of DO depletion. The algal bloom may suffice to raise BOD so that DO is depleted. Because no chlorophyll *a* or algal biomass data were collected in this study, the cause of BOD to the river can only be estimated from BOD, measured at several points, and COD (chemical oxygen demand) measured at point sources such as the WWTP above RM 5.4 in 1998.

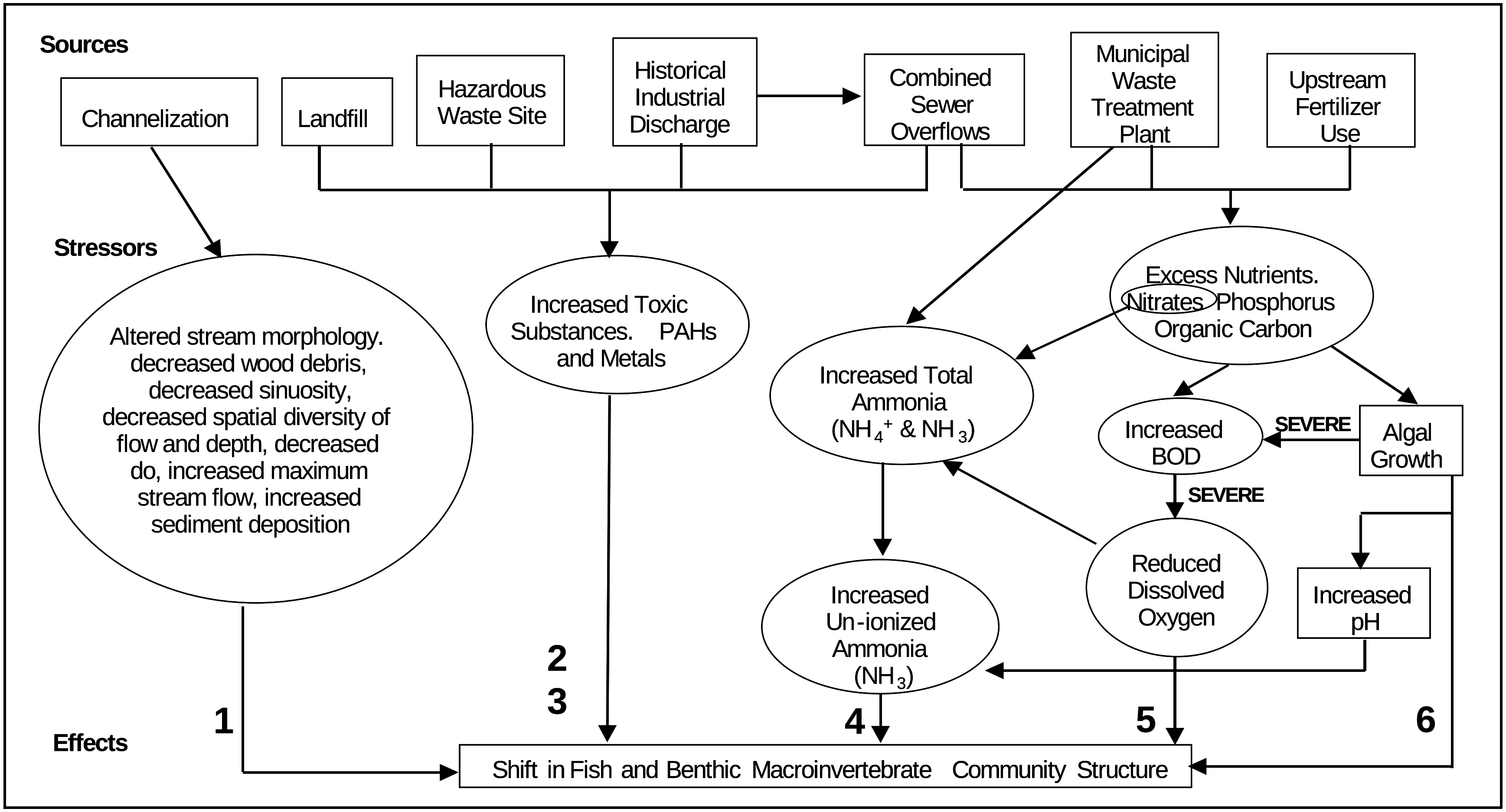
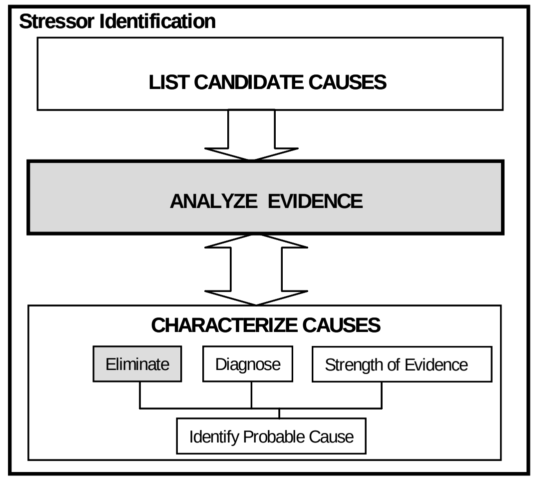


Figure 7-4. A conceptual model of the six candidate causes for the Little Scioto stressor identification. (Potential sources are listed in top most rectangles. Potential stressors and interactions are located in ovals. Candidate causes are numbered 1 through 6. Note that some causes have more than one stressor or more than one step associated with it. The impairments are located in the lower rectangle.)

6. Nutrient Enrichment - The sixth and final candidate cause is a less extreme form of nutrient enrichment. Primary production and organic matter loading to the sediments are increased, but not enough to reduce DO. This can cause changes in fish and benthic macroinvertebrate assemblages, including changes in dominant species, and greatly increased abundance and biomass (Carpenter et al. 1988, Rankin et al.1999, Smith et al. 1999, Dodds and Welsh 2000, Edwards et al. 2000). This form of nutrient enrichment is also associated with fin erosion (Rankin et al. 1999).

7.5 Analyze Evidence to Eliminate Alternatives

7.5.1 Data Analyzed



Habitat alteration-related data

Data on the spatial location of habitat alteration was obtained by using the Qualitative Habitat Evaluation Index (QHEI). The QHEI incorporates measures of habitat condition and has been correlated with the IBI. This index uses eight interrelated metrics, which assess substrate type and quality; in-stream cover type and amount; channel morphology; riparian width and quality and bank erosion; pool / riffle characteristics including depth, current, pool morphology, substrate stability and riffle embeddedness; and finally gradient (Rankin 1989). Based on these metrics, a total score is assigned to a stream reach out of a possible 100 points, with greater scores indicating higher quality. The channel morphology and substrate metrics are particularly relevant for this case because of the channelization (Figure 7-5). Values for the QHEI and its component metrics are given in Table 7-15 (see Section 7.13).

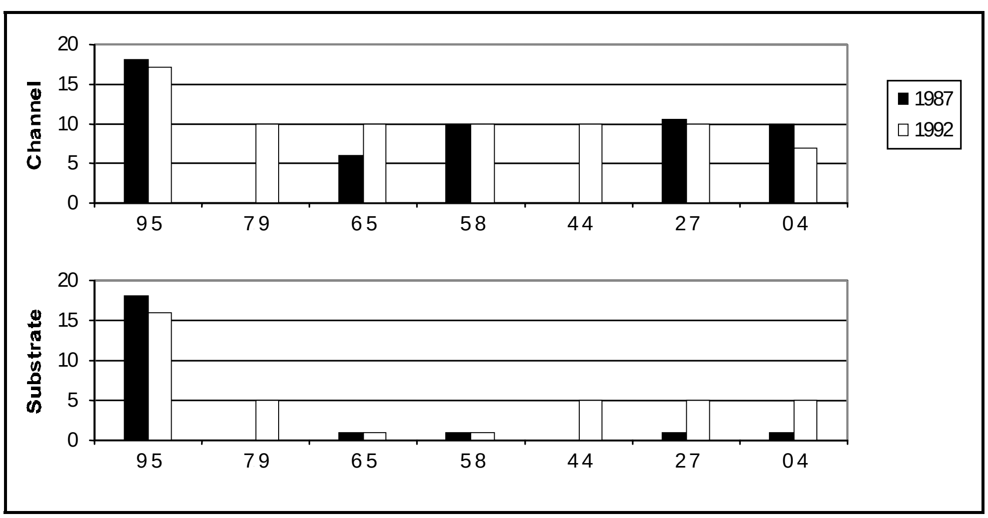


Figure 7-5. Selected QHEI metrics for 1987 and 1992. (Scores are qualitative ranks.)

Chemical Data
Data on sediment and in-stream chemistry were used to evaluate the spatial location of the remaining candidate causes (#2-6). Nutrient concentrations measured in water included ammonia, nitrates and nitrites (NO_X), phosphorus (P), and BOD. Ambient levels of potential toxic chemicals were determined for sediment and water. Results of chemical analyses are presented in Tables 7-16, 7-17 and 7-18 (See Section 7.13), and Figures 7-6, 7-7, and 7-8.

While PAHs were not detectable at the upstream sites (RM 9.5 and 7.9), many PAHs were detected between RM 6.5 and 0.4 (Table 7-16) (Figure 7-6). Spearman Rank Correlations between chemical and biological data from 1992 at RM 5.7 to 0.4 are shown in Table 7-2 through 7-5.

Metals were found in sediments at relatively high concentrations at RM 6.5 and downstream (Table 7-17; see Section 7.13) (Figure 7-7). These included lead, cadmium, copper, chromium, zinc, and mercury. Arsenic was relatively high at upstream reference and study sites. Spearman rank correlations between metals and biological data from 1992 at RM 5.8 to 0.4 are shown in Table 7-3. Strong correlations having the sign that is consistent with the hypothesis were noted for copper and mercury.

The water quality parameters ammonia, nitrates and nitrites (NO_x), and BOD increased substantially at RM 5.8, and remained elevated. Dissolved oxygen declined at 7.9 and remained low to RM 0.4 (Table 7-18; see Section 7.13) (Figure 7-8). Spearman rank correlations of water chemistry and biological endpoints are presented in Table 7-4. Percent Tanytarsini are significantly correlated with DO, BOD, NO_x and P, and the negative direction of the slope was consistent with ecological theory. Percent DELTA was correlated with the same parameters (DO, BOD, NO_x and P) but at the 0.8 level, whereas percent Cricotopus was associated with ammonia and the QHEI.

7.5.2 Associations between Candidate Causes and Effects

The associations between candidate causes and effects were analyzed by combining data on the location of the three impairments with data on habitat quality and chemical concentrations in water and sediments. The analyses evaluated whether the candidate causes and each of the three impairments were spatially co-located, and whether a gradient in recovery corresponded with a decrease in the candidate cause. These associations are organized in table format (Table 7-5).

The first objective of the analysis was to determine if there was evidence that the candidate cause occurred at the same place as the impairment but not where that particular impairment was absent. Plots of the channel quality and substrate metrics from the QHEI are shown in Figure 7-5. The chemistry values relevant to each of the causal scenarios are shown in Figures 7-6, 7-7 and 7-8. Each graph shows the level or concentration of the parameter. The presence or absence of candidate causes at the locations of Impairments A, B, and C are summarized in Table 7-5.

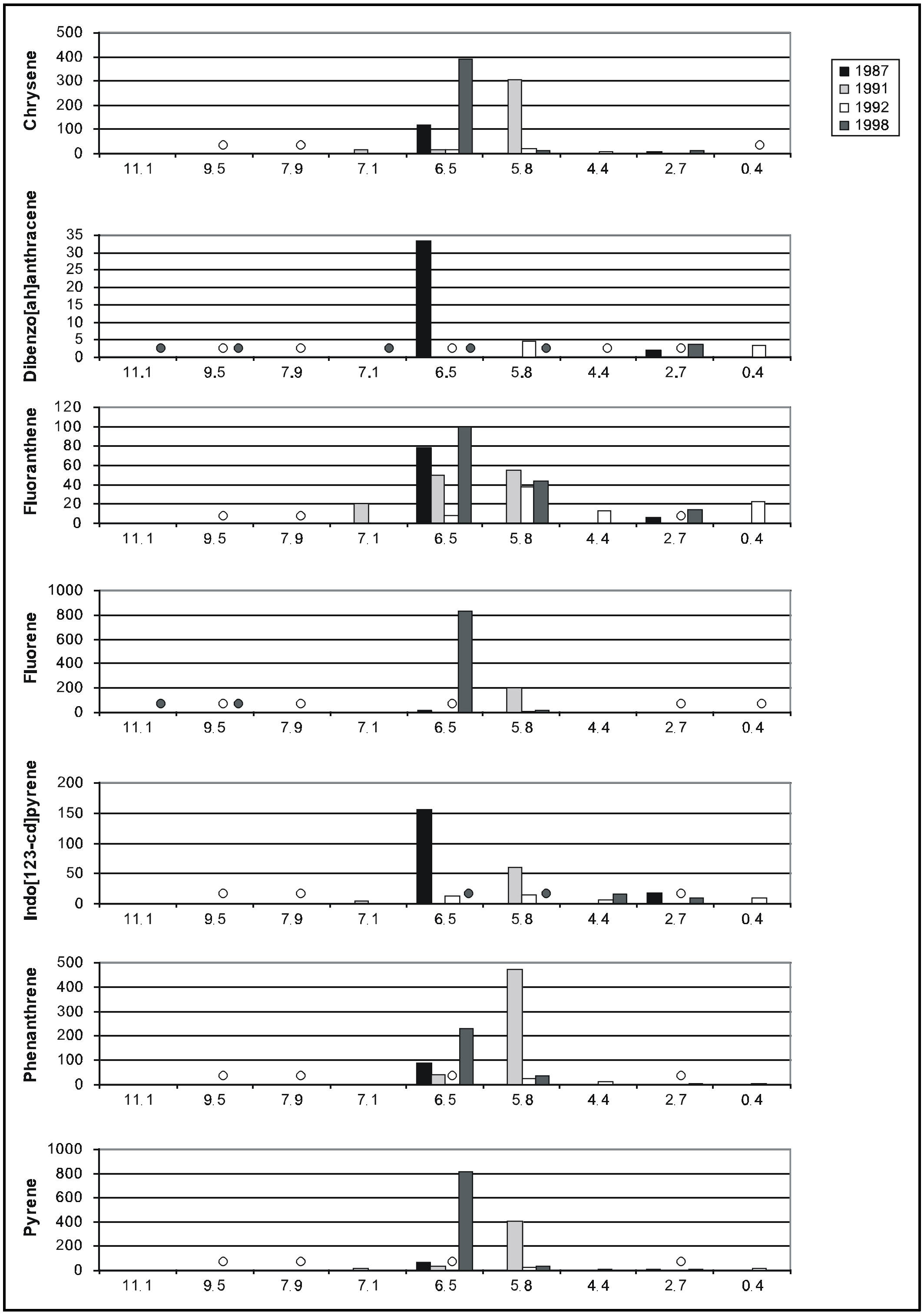


Figure 7-6. Mean PAH concentrations from the sediment (mg/kg) in the Little Scioto River 1987-1998. ((o) indicates below detection limit. Absence of bar indicates no data available.)

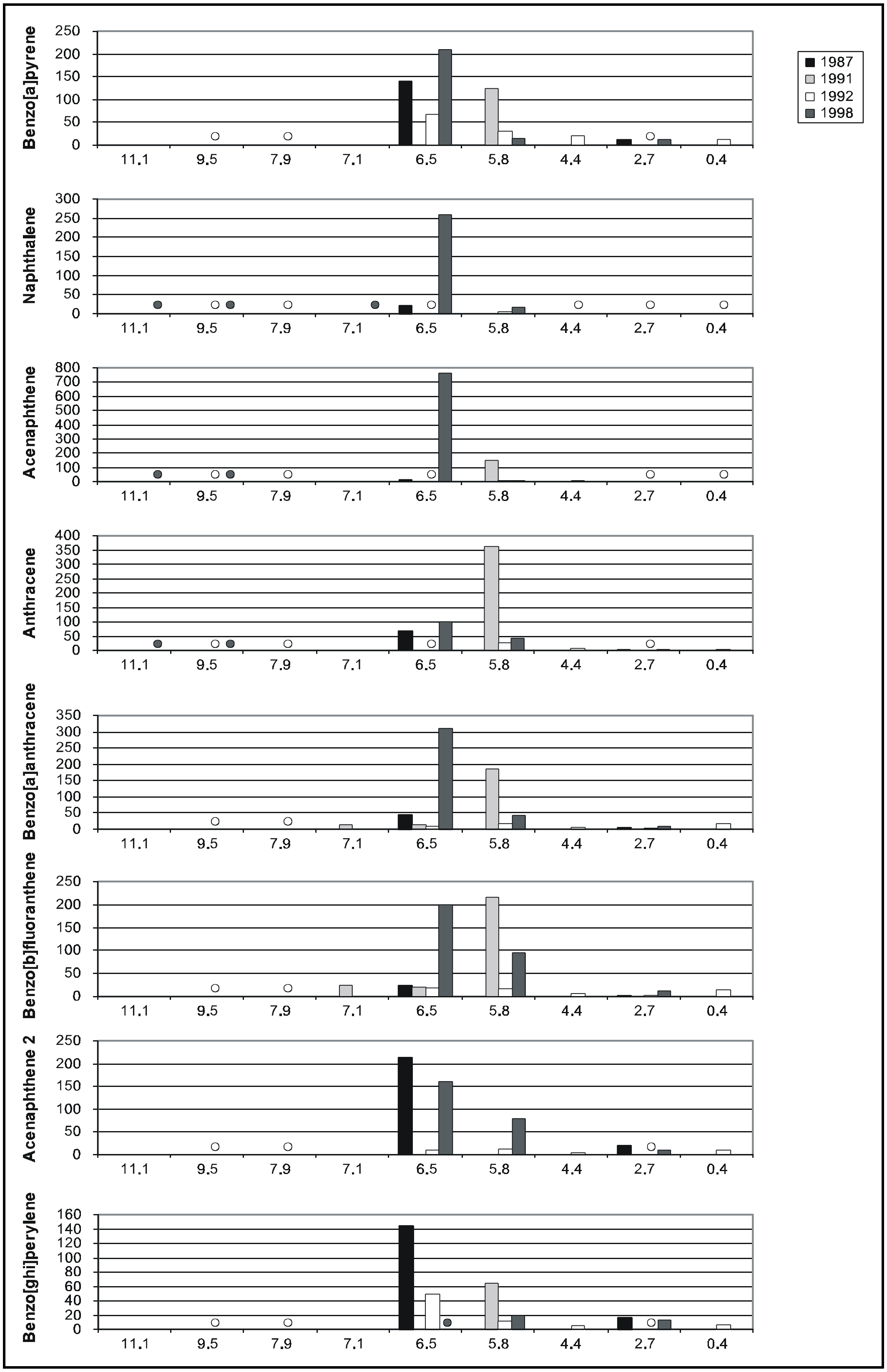


Figure 7-6 (continued). Mean PAH concentrations from the sediment (mg/kg) in the Little Scioto River 1987-1998. ((o) indicates below detection limit. Absence of bar indicates no data available.)

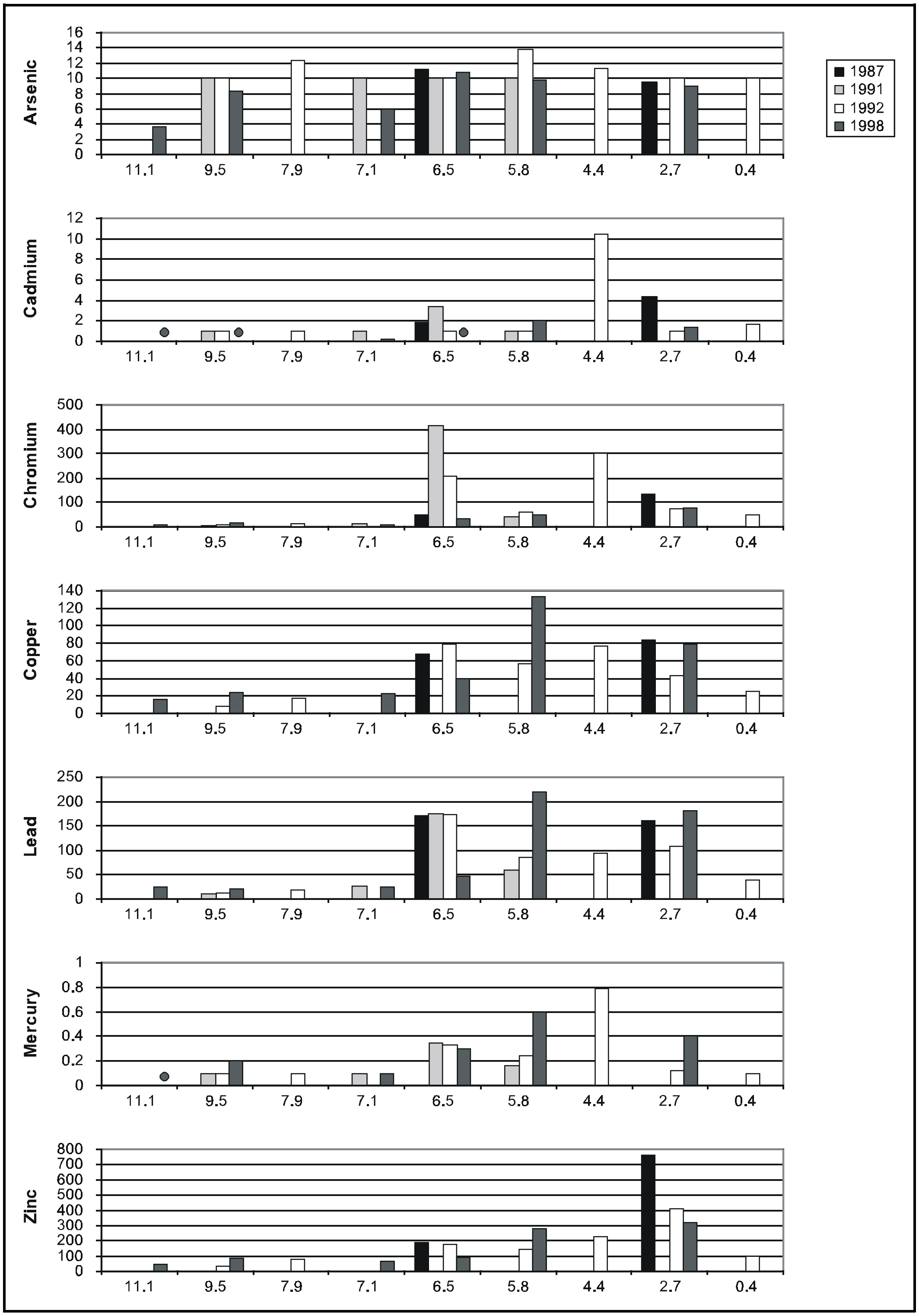


Figure 7-7. Mean metal concentrations from the sediment (mg/kg) in the Little Scioto River from 1987-1998. (Absence of bar indicates no data available.)

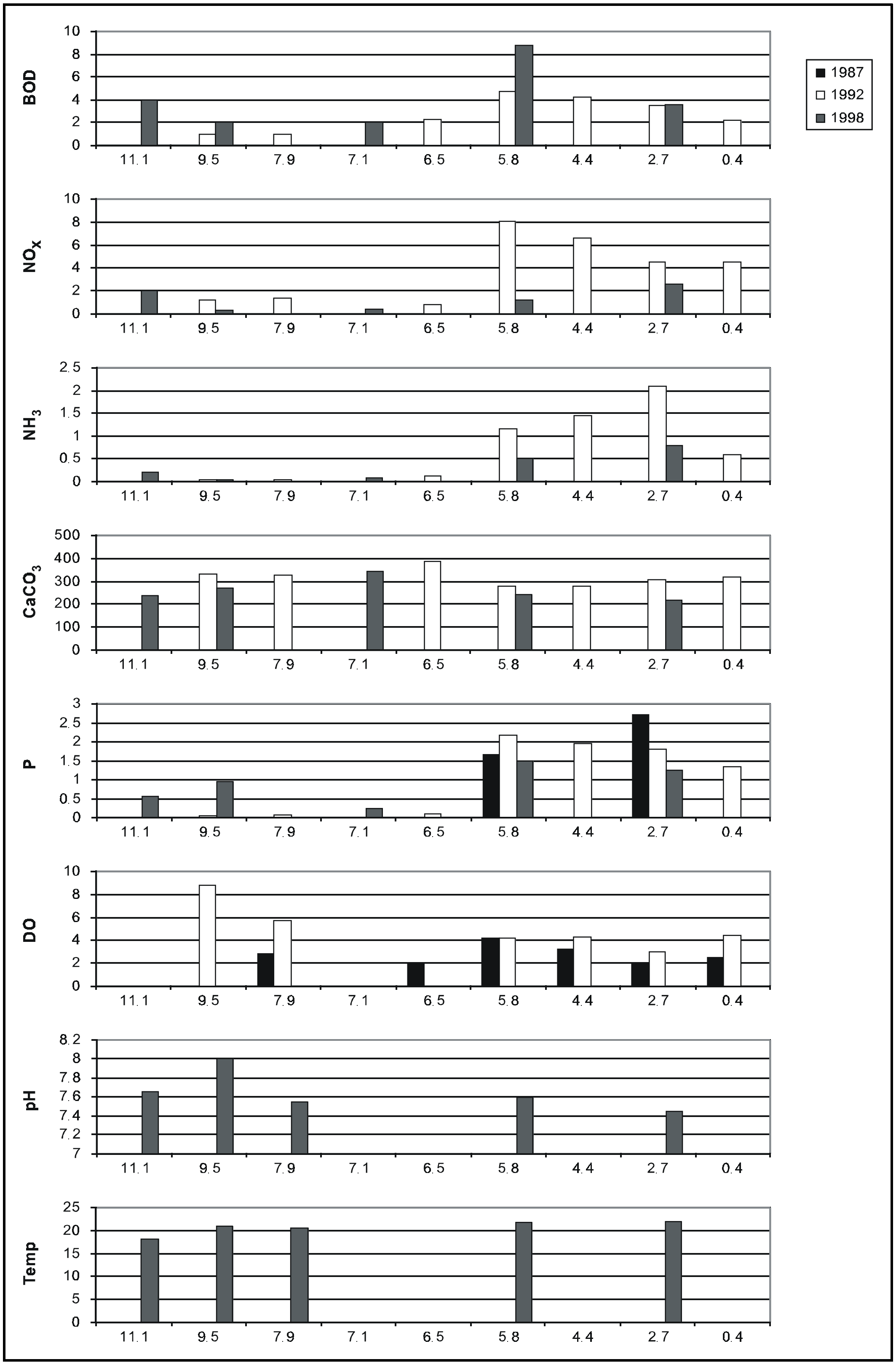


Figure 7-8. Mean water chemistry values from the Little Scioto River from 1987-1998. (BOD, NO_x, Ammonia, $CaCO_3$, PO_4, are all mg/L, Temperature (^{0}C). DO is also mg/L and is the minimum value obtained from grab samples for each year. Absence of a bar indicates no data for that year.)

Table 7-2. Spearman rank correlations with selected metrics and the IBI and ICI from 1992 and selected PAHs. (Reflects only values from RM 5.8 to 0.4. Correlations N=4).

Parameter	DELTA	% Tanytarsini Midges	% Cricotopus
Anthracene (#2)	0.60	-0.74	-0.20
Benzo[a]anthracene (#2)	0.00	-0.21	-0.40
Benzo[ghi]perylene(#2)	0.00	-0.21	-0.40
Benzo[a]pyrene (#2)	0.00	-0.21	-0.40
Chrysene (#2)	0.80*	-0.95*	0.40
Dibenzo[a,h]anthracene (#2)	-0.21	-0.06	-0.21
Fluoranthene (#2)	0.00	-0.21	-0.40
Fluorene (#2)	0.74	-0.89*	0.11
Naphthalene (#2)	0.26	-0.54	0.26
Phenanthrene (#2)	0.60	-0.74	-0.20
Pyrene (#2)	0.00	-0.21	-0.40

* Correlations above 0.8

Table 7-3. Spearman rank correlations with selected metrics and the IBI and ICI from 1992 and selected metals. (Reflects only values from RM 5.8 to 0.4. Correlations N=4).

Parameter (Candidate Cause)	DELTA	% Tanytarsini Midges	% Cricotopus
Arsenic (#3)	0.74	-0.89*	0.11
Cadmium (#3)	0.20	0.11	-0.60
Chromium (#3)	0.80*	-0.63	0.40
Copper (#3)	1.00*	-0.95*	0.20
Lead (#3)	0.40	-0.32	0.80*
Mercury (#3)	1.00*	-0.95*	0.20
Zinc (#3)	0.40	-0.32	0.80*

* Correlations above 0.8

Table 7-4. Spearman rank correlations with selected metrics and the IBI and ICI from 1992 and selected water quality and habitat quality measurements. (Reflects only values from RM 5.8 to 0.4. Correlations N=4).

Parameter (Candidate Cause)	DELTA	% Tanytarsini Midges	% Cricotopus
Channel Metric (#1)	0.77	-0.82*	0.77
QHEI (#1)	0.20	-0.32	1.00*
Ammonia, N (#4)	0.40	-0.32	0.80*
Dissolved oxygen maximum (#5)	0.80*	-0.95*	0.40
Dissolved oxygen minimum (#5)	0.60	-0.74	-0.20
BOD (#5)	0.80*	-0.95*	0.40
Nitrate-nitrite, N (#4,5,6)	0.80*	-0.95*	0.40
Phosphorus, total P (#5,6)	0.80*	-0.95*	0.40

* Correlations above 0.8

The second objective was to determine if the cause increased compared to the nearest upstream location. Statistical analyses were not used to determine an increase because the power would be very weak due to small sample sizes. Even a small increase was accepted since it might represent a threshold for the effect (Table 7-5).

The third objective of the analyses was to evaluate whether a gradient in the intensity of the potential cause corresponded to a gradient of recovery in impairment. The gradient analysis was conducted only for Impairment C, which was observed at four contiguous locations (i.e., RM 5.8 to 0.4). The recovery of Impairment B could not be analyzed since it would be masked by Impairment C. Similarly, any recovery of Impairment A would be masked by both B and C. The gradients in environmental parameters and the IBI and ICI were examined visually by comparing Figures 7-2 and 7-3 with Figures 7-5 through 7-8. The IBI and ICI metrics for 1987 and 1992 data are shown in Table 7-13 and Table 7-14, respectively. In addition, Spearman's rank correlations were calculated using the 1992 data set to relate the biological metrics (shown in Figure 7-3) with each of the parameters related to the candidate causes. The results of this analysis are shown in Tables 7-2 through 7-4.

Two metrics are more severe at Impairment C: % DELTA and % Tanytarsini midges decrease and % *Cricotopus* increases. Percent DELTA were significantly correlated with copper and mercury, and moderately correlated with chrysene, chromium, BOD, nitrate, phosphorous, and maximum DO. The change in tanytarsini midges was negatively and strongly correlated with chrysene, copper, mercury, BOD, nitrate, phosphorous, maximum dissolved oxygen, and moderately correlated with fluorene, arsenic, and the channel metric. The change in % *Cricotopus* was strongly positively correlated with QHEI and moderately correlated with lead, zinc, and ammonia.

Table 7-5. Evidence for eliminating candidates causes at Impairments A, B, and C.

	Impairment A	**Impairment B**	**Impairment C**
Habitat Alteration (Candidate Cause 1)			
Is there exposure at the same location as the impairment?	Yes	Yes	Yes
Is exposure increased over the closest upstream location?	Yes	No	No
Is there a gradient of recovery as exposure decreases?	NA* (Gradient in impairment is masked by B and C)	NA (Gradient in impairment is masked by C)	No (Correlation coefficients have the wrong signs, with % DELTA and % Tanytarsini)
Is the exposure pathway complete?	Yes	Yes	Yes
PAH Contamination (Candidate Cause 2)			
Is there exposure at the same location as the impairment?	No	Yes	Yes
Is exposure increased over the closest upstream location?	No	Yes	No (based on metabolite values in fish)
Is there a gradient of recovery as exposure decreases?	NA (Gradient in impairment is masked by B and C)	NA (Gradient in impairment is masked by C)	Inconclusive (Mixed results)
Is the exposure pathway complete?	No	Yes	Yes

Table 7-5 (continued). Evidence for eliminating candidates causes at Impairments A, B, and C.

	Impairment A	Impairment B	Impairment C
Metal Contamination (Candidate Cause 3)			
Is there exposure at the same location as the impairment?	Yes	Yes	Yes
Is exposure increased over the closest upstream location?	Yes (all metals greater in some years)	Yes (all metals greater)	Yes (copper and zinc increased)
Is there a gradient of recovery as exposure decreases?	NA (Gradient in impairment is masked by B and C)	NA (Gradient in impairment is masked by C)	Yes (Tanytarsini midges and % DELTA are strongly correlated with copper and mercury)
Is the exposure pathway complete?	Yes	Yes	Yes
Ammonia (Candidate Cause 4)			
Is there exposure at the same location as the impairment?	Yes	Yes	Yes
Is exposure increased over the closest upstream location?	No	Yes	Yes
Is there a gradient of recovery as exposure decreases?	NA (Gradient in impairment is masked by B and C)	NA (Gradient in impairment is masked by C)	NA (ammonia increases below RM 5.8)
Is the exposure pathway complete?	No	Yes	Yes

Table 7-5 (continued). Evidence for eliminating candidates causes at Impairments A, B, and C.

	Impairment A	Impairment B	Impairment C
Low Dissolved Oxygen/High BOD (Candidate Cause 5)			
Is there exposure at the same location as the impairment?	Yes	Yes	Yes
Is exposure increased over the closest upstream location?	No (DO is depressed, BOD unchanged)	Yes (BOD is two times greater in 1992, DO slightly less)	No (BOD is elevated, but DO is greater than either RM 7.9 or RM 6.5)
Is there a gradient of recovery as exposure decreases?	NA (Gradient in impairment is masked by B and C)	NA (Gradient in impairment is masked by C)	NA (ammonia increases below RM 5.8)
Is the exposure pathway complete?	Yes	Yes	No
Nutrient Enrichment (Candidate Cause 6)			
Is there exposure at the same location as the impairment?	Yes	Yes	Yes
Is exposure increased over the closest upstream location?	Yes	Yes	Yes
Is there a gradient of recovery as exposure decreases?	NA (Gradient in impairment is masked by B and C)	NA (Gradient in impairment is masked by C)	Yes (% Tanytarsini and % DELTA are strongly correlated with NO_x and Total P)
Is the exposure pathway complete?	Yes	Yes	Yes

NA* = not applicable

7.5.3 Measurements Associated with the Causal Mechanism: Exposure Pathways

The exposure pathways are shown in Figure 7-4. Lines of evidence for each exposure pathway are discussed below and are summarized in Table 7-11. To refute an hypothesis, a step in the pathway must be absent.

Habitat Alteration (#1) - Channelization results in a constellation of stressors, including loss of riffles with increased sediment deposition, and decreased DO. The QHEI metrics can yield insights into specific changes: for example, riffle scores are zero throughout the channelized portion of the stream (Table 7-15; see Section 7.13), substrate quality and embeddedness due to fine sediment drops at RM 7.9, and DO also drops at RM 7.9. The co-occurrence of macroinvertebrates with changes in physical structure may be somewhat lessened, because Hester-Dendy samplers create an artificial solid substrate for colonization. The ICI score does include a qualitative kick net sample that is independent of the artificial substrates. The exposure pathway for habitat alteration is complete for Impairments A, B, and C.

PAHs (#2) - Exposure to PAHs involves two steps: direct contact with external tissues and uptake into the organism. Because the PAH information in this case is from the sediments, we assume that fish and benthic invertebrates between river miles 6.5 and 0.4 will contact this contamination. Concentrations of PAH in the sediment were used only from samples collected in 1992, as it was the only year in which we were confident that the samples were collected from the top six inches. It is unlikely that fish or invertebrates would be exposed to deeper sediments.

The exposure pathway for PAHs could be interrupted if there was no sign of internal exposure. Aquatic contaminants such as PAHs have been monitored by measuring the metabolites of xenobiotics in fish bile (Roubal et al. 1977, Gmur and Varanasi 1982, Varanasi et al. 1983). Samples from white suckers (*Catostomus commersoni*) taken in 1992 from the Little Scioto River were analyzed for concentrations of benzo[a]pyrene (BAP) and naphthalene (NAPH)-type metabolites. Results of the analysis of PAH bile metabolites in white suckers from the Little Scioto River are shown in Figure 7-9. Biomarkers of NAPH and BAP are elevated from RM 6.5 to the mouth of the river, providing evidence that the exposure pathway is complete at these locations. Exposure criteria, concentrations considered to be above background, were exceeded at RMs 6.5 through 0.4. PAHs are also known to cause induction of detoxifying enzymes such as EROD. EROD activity was elevated at RM 6.5 - 0.4. Based on the absence or presence of bile metabolites, the exposure pathway for PAHs is incomplete at Impairment A, and complete at Impairments B and C.

Metals (#3) - Metals must be taken into organisms to cause adverse effects. Data from fish tissue sampled in 1992 confirm uptake of lead and zinc. For common carp (*Cyprinus carpio)* at RM 9.2, zinc concentrations were 79.6 mg/kg, at RM 6.5, zinc concentrations were 68.3 mg/kg. For white suckers at RM 6.5, zinc concentrations were 17.8 mg/kg, and lead concentrations were 81.4 mg/kg. At RM 2.7, fish tissues levels were 15.8 mg/kg for zinc and 0.34 mg/kg for lead. For the other metals, we have conservatively assumed that external exposure will represent internal exposure for fish. Making this assumption, increased exposure to at least one of the metals occurs at all sampling locations in the reach RM 7.9 to 0.4. Concentrations of metals in sediment were from samples taken in 1992 from the top six inches of sediment. For 1987 and 1998 data, the depth of samples is unknown.

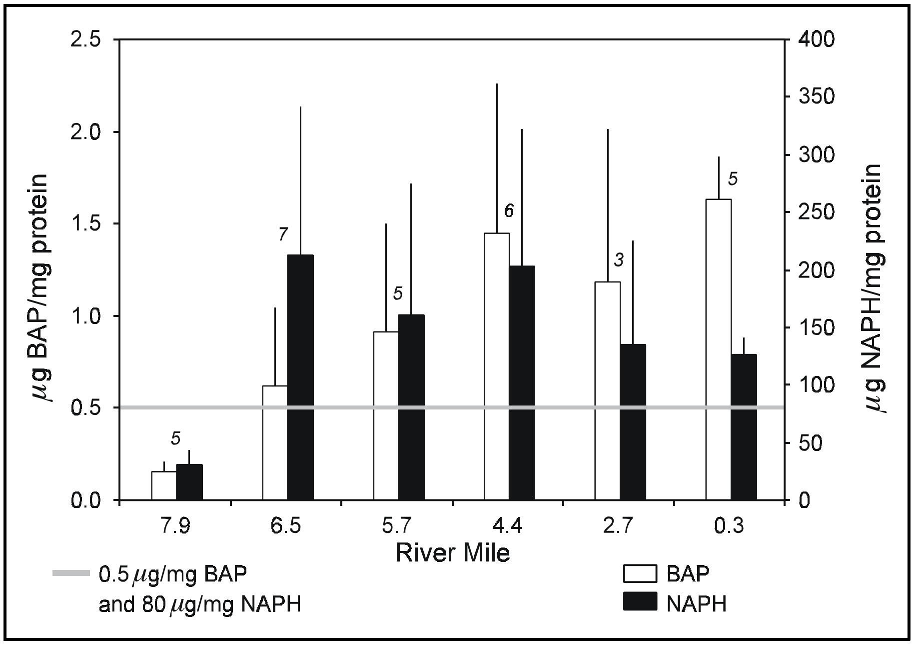


Figure. 7-9. Bile metabolites (µg/mg protein) measured in white suckers from the Little Scioto River in 1992. (Median levels of PAH metabolites below RM 7.9 were up as much as 4 times the Exposure Criteria, (dashed horizontal line) which are upper limits of background for the state of Ohio. The numbers above the bars equal number of fish sampled. Vertical lines are standard errors.)

Ammonia (#4) - There are several interweaving pathways by which ammonia can be produced in the river and cause effects. We have evidence for two of these steps: total ammonia, and nitrate and nitrite concentrations that are converted to ammonia when DO is low. Toxic unionized ammonia is formed at high pH. Hard water streams of the Eastern Corn Belt Plains typically have pH from 7.5-8; pH may rise even above 9.0 in the summer during maximum photosynthesis in nutrient-enriched waters. Data on pH are not available in 1992, however, in 1998 grab samples, pH ranged between 7.4 to 8.0. The Little Scioto is highly enriched, and it is highly likely that there are periods when pH is greater than indicated by grab samples. Thus, we assume that the exposure pathway is complete in the Little Scioto when total ammonia is present. This occurs from RM 11.1 to 0.4. Because ammonia concentrations are measured in the water column, both fish and macroinvertebrates are exposed.

Low Dissolved Oxygen/High Biological Oxygen Demand (#5) - Dissolved oxygen can be depleted by high BOD due to the bacterial respiration associated with allochthonous organic matter or decaying algal mats. We have measurements of several relevant parameters: NO_x, total P, BOD and DO concentrations. This exposure pathway is considered complete under two scenarios: (1) BOD is elevated and DO is reduced compared with the most upstream location, or (2) if BOD data is unavailable, NO_x and P are elevated and assumed to cause algal growth, and DO is reduced as compared with the most upstream location. At RM 7.9, DO is reduced, but BOD is unchanged, so that the exposure pathway is considered incomplete. RM 6.5 is more difficult to evaluate because data are scanty and are used from different years. In 1987, DO data were low at

RM 6.5, and in 1992, the BOD was slightly elevated; thus, the pathway is complete. At RM 5.8 to 0.4, because BOD is elevated but DO is similar or greater than at 7.9, the exposure pathway is considered incomplete.

Nutrient Enrichment (#6) - We have evidence for the presence of elevated levels of both NO_x and total P concentrations. This exposure pathway appears to be complete at RM 5.8 to 0.4, and at RM 7.9.

7.5.4 Summary of Analyses for Elimination

The results of the analysis of spatial associations are summarized in Table 7-5 (pages 7-21 to 7-23). The table addresses four questions for each combination of impairment and candidate cause. If any of the answers are no, then the candidate cause can be eliminated:

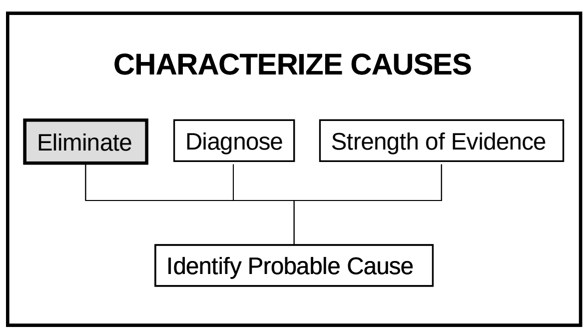
- The first question is whether a candidate cause and impairment are spatially co-located. Regardless of concentration, the answer is yes if the stressor is present. If the stressor is not present, the answer is no and the impairment could not have been caused by exposure to that stressor.

- The second question asks whether the exposure is elevated compared to the closest upstream location where the impairment does not occur. The candidate cause could have been responsible for the impairment only if exposure increased. The candidate cause can be eliminated if the answer to the second question is no.

- The third question asks whether there is a decrease in exposure that corresponds with recovery of the impairment. As discussed above, this question is relevant only to Impairment C. If the answer is no with results clearly showing a lessening of impairment with consistent exposure, then the candidate cause can be eliminated.

- The last question asks if the exposure pathway is complete. If it is interrupted or clearly incomplete so that exposure could not have taken place, then it can be eliminated as a potential cause.

7.6 Characterize Causes: Eliminate



Potential causes may be eliminated if the evidence indicates that they do not co-occur with effects, if effects decrease with increasing influence of the cause, or if the exposure pathway is incomplete. Each of the three Impairments (A, B, and C) are discussed below in relation to the elimination of specific causes. Conclusions about which candidate causes remain for each impairment are also listed.

Impairment A: RM 7.9

Habitat alteration and metal contamination are the only candidate causes known to co-occur at RM 7.9 and to increase compared to upstream locations. All metals were slightly greater at RM 7.9 compared to RM 9.2. PAHs and ammonia were not elevated at RM 7.9 relative to the upstream reference, thus candidate causes #2 and

#4 are eliminated. DO concentrations were about 30% lower than upstream, but BOD concentrations were not different from the upstream reference location (RM 9.2), thus candidate cause #5 is eliminated. NO_x increased from 1.2 mg/L to 1.4 mg/L. The shift is small, but precludes elimination of candidate cause #6.

Conclusion: Habitat Alteration (#1), Metal Contamination (#3), and Nutrient Enrichment (#6) remain.

Impairment B: RM 6.5.

At this site, only candidate cause #1 can be eliminated because the degree of habitat alteration is not elevated compared with those at RM 7.9. The decline in QHEI score is associated with the obvious presence of organic chemical contamination rather than physical stream characteristics. Organic chemicals, including benzo[a]pyrene and naphthalene, were present and were elevated above concentrations at RM 7.9. Exposure to these organic chemicals was demonstrated by internal concentrations of metabolites. The metals chromium, copper, lead, and mercury were elevated compared to upstream concentrations in all years for which there is data, including 1988, 1991, 1992 and 1998. Dissolved oxygen levels were among the lowest in the river in 1987, and BOD levels were slightly greater than upstream locations. Ammonia concentrations were also slightly greater, and total P concentrations were 0.02 mg/L greater.

Conclusion: PAH Contamination (#2) Metal Contamination (#3), Ammonia Toxicity (#4), Low Dissolved Oxygen/High Biological Oxygen Demand (#5), and Nutrient Enrichment (#6) remain.

Impairment C: RM 5.7.

In this reach of the river, the degree of habitat alteration and PAH levels were similar or lower than at RM 6.5, thus candidates #1 and #2 are eliminated. Candidate cause #5, low DO/high BOD, can be eliminated, even though BOD, P and NO_x are elevated because the subsequent event in the pathway, decreased DO, did not occur. DO is unchanged from RM 7.9 in 1992, and RM 6.5 in 1987. The metals (copper and zinc) increased slightly, and the copper gradient was significantly correlated with % Tanytarsini midges and % DELTA, thus candidate cause #3 remains. NO_x and P were elevated in the reach compared to upstream locations and were significantly correlated with % Tanytarsini midges. Candidate cause #6 remains. Candidate cause #4 could be eliminated since ammonia was not correlated with the specific impairments. However, the increase in ammonia was 10 times greater than upstream, and because the data available for correlations were very limited, a conservative decision could be made to retain this cause for further evaluation by the strength of evidence approach.

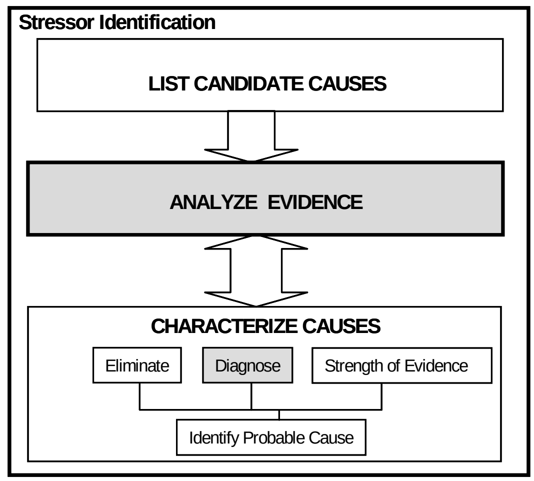
Conclusion: Metal Contamination (#3), Ammonia (#4), and Nutrient Enrichment (#6) remain.

A summary of the candidate causes that remain after the elimination process are listed in Table 7-6. Only those causes remaining need to be evaluated by diagnostic or strength of evidence analyses.

Table 7-6. Candidate causes remaining after elimination.

	Impairment A	Impairment B	Impairment C
#1 Habitat alteration	X		
#2 PAH Contamination		X	
#3 Metal Contamination	X	X	X
#4 Ammonia		X	X
#5 Low DO/BOD		X	
#6 Nutrient Enrichment	X	X	X

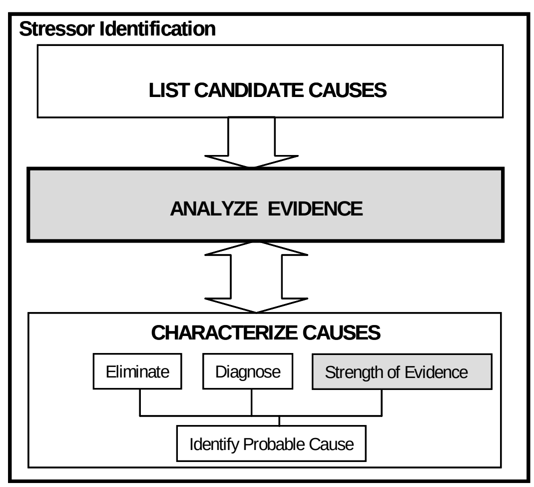
7.7 Analyze Evidence for Diagnosis



Diagnosis is the identification of causes based on characteristic signs or symptoms (see 4.2.2). No evidence strong enough to support diagnosis was available for any of the candidate stressors. However, the pattern of community change is considered to be suggestive, and is used in the strength of evidence analysis below.

The deformities, fin erosion, tumors, physical lesions and anomalies on fish that constitute the DELTA are pathologies that are also potentially subject to diagnosis. Some DELTA are strongly associated with known toxic substances and others with increased nutrients (Yoder and Rankin 1995b). However, no pathologist has examined the fish in question. DELTA cannot be used to distinguish among toxic substances unless specific anomalies are identified, and even these may be too non-specific to diagnose without additional information (e.g., histopathology).

7.8 Analyze Evidence to Compare Strength of Evidence



All of the remaining candidate causes are subjected to a strength of evidence analysis to verify the elimination step and to identify the most likely cause from the multiple hypothesized causes that remained after the elimination process. The strength of evidence analysis examined case specific evidence as well as evidence from other situation and biological knowledge.

<u>Case Specific Evidence</u>

The evidence presented earlier for the elimination step is useful here as well. In addition, some data on loadings are available from the Waste Water Treatment Plant (WWTP), which discharges at RM

6.2 and also has combined sewer overflows that discharge during wet weather periods. No clear trends were evident in the loadings of total non-filterable residue or biological oxygen demand between 1977 and 1992. Ammonia values were generally low, with fifty percent of loading below 10 kg/day between 1977 and 1991. The highest ammonia loading occurred during 1992, with a median of 12.6 kg/day and a maximum of 130 kg/day (OEPA 1994).

Evidence from Other Situations or Biological Knowledge

This section presents evidence that uses information from other studies that are related to either exposures or effects found in segments of the Little Scioto River. In particular, associations are made between the exposures known at the site and reports of effects caused by similar exposures. This section also uses levels of effects seen at the site and effects seen at other sites where the same candidate cause occurred. It also considers special experimental evidence; that is, reports about places with similar stressors and effects that improved when the stressor was removed, and laboratory studies of candidate cause-effect relationships.

Exposure-response data are available for PAHs and metals, although not for the community parameters of greatest interest for this study. Sediment effect concentrations (SECs) developed for *Hyalella azteca* and *Chironomus riparius* were considered, but only *Hyalella azteca* was used since *Chironomus riparius* values were always less sensitive. Sediment effect concentrations for *Hyalella azteca* are expressed as threshold effect level (TEL) and probable effect level (PEL) (Table 7-19; see Section 7.13) (USEPA 1996b).

The TEL and PEL are sediment concentrations associated with toxicity in laboratory tests. The interpretation is that toxicity rarely occurs below the TEL and frequently occurs above the PEL (USEPA 1996b). Values were derived from a data set consisting of many similar studies, and they consider both effect and no-effect data for field-contaminated sediments. The TEL and PEL values used in this study are listed in Tables 7-19 and 7-20 (see Section 7.13). Since many metals and PAHs were present at sites, partial toxicity contributed by individual chemicals were calculated and summed to estimate the overall toxicity of metals and PAH at each site. TELs and PELs are used with caution because they are based on sediments with multiple contaminants.

The TEL and PEL values were compared with the concentrations seen at the locations of impairment in Table 7-19. As shown in Table 7-19, the most striking result is that no PAH exceeded any criterion level at Impairment A for 1992. For metals only, the TEL for arsenic was exceeded at Impairment A in 1992. At Impairment B and C, the *Hyalella azteca* PEL and TEL were exceeded for all PAH that were measured and in every year except 1992, when there were more samples below the detection limit. *Hyalella azteca* TEL values were exceeded for most metals, but only a few PEL values were exceeded, including those for lead, copper, and chromium.

For PAHs, the cumulative toxic units were exceeded at Impairments B and C in every year (Table 7-7). Exceedances ranged from 339 to 18,820 times the value that would probably kill *Hyalella azteca*. For metals, the cumulative toxic units were also exceeded at Impairments B and C in every year. However, exceedances were never more than six times the cumulative probable effect level.

Table 7-7. Cumulative toxic units for PAHs and metals based on the PEL values. (Values greater than 1.0 exceed PEL*).

Chemical	Cumulative Toxic Units			
	Nearest Upstream Location	Impairment A	Impairment B	Impairment C
PAH	/0\ (0) [1.2]*	/0\ (0) [2.5]*	/604.5* (339.4)* [18819.9]*	/9697.8* (821)* [1633.4]*
Metals	/0.4\ (0.6) [0.9]	/0.7\ (1.1)* [0.9]	/4.3* (5.1)* [1.6]*	/1.5* (2.8)* [5.8]*

* Exceeds PEL and TEL. / \ = 1987-1991, ( ) = 1992, [ ] = 1998. Zero = below detection.

Criteria are also available for ammonia (USEPA 1998b) (Table 7-8). The toxicity of total ammonia (which includes NH_3 and NH_4^+) varies with pH. Dehydration of ammonium ion (NH_4^+) to un-ionized ammonia is controlled by ambient pH, such that excess hydroxide ions (high pH) increase the concentration of the more toxic, un-ionized form. Hard water streams of the Eastern Corn Belt Plains (ECBP) typically have pH from 7.5-8; in the summer, during maximum photosynthesis in nutrient enriched waters, pH may rise above 9.0. In 1998, pH values ranged between 7.4 and 8.4, and appeared to be independent of location. Total ammonia concentrations at RM 5.8 through 2.7 would have exceeded the ammonia criterion for water having a pH 8.0 to 8.5 in 1992 (Table 7-8). In 1998, the criterion would have been exceeded at pH 8.5.

Ohio's criteria for dissolved oxygen (causal candidate #5) are 4.0 mg/l for warm water, and 3.0 mg/l for modified warm water. In 1992, no locations had dissolved oxygen below the modified warm water criterion, and only RM 2.7 had dissolved oxygen concentrations below the warm water criterion, based on a single measurement. However, in 1987, continuous data were collected by Datasonde (in-stream Hydrolab) and violations were detected at Impairments A and B (Table 7-8).

Ohio's proposed state-wide criterion for modified warm-water habitat for nitrate and nitrite is 1.6 mg/L for wadeable streams in the ECBP having a drainage greater than 20 mi^2 and less than 200 mi^2. For total phosphorus, the proposed state-wide criterion for modified warm-water habitat is 0.28 mg/L (Rankin et al. 1999). These are exceeded at RM 5.8 (Table 7-8).

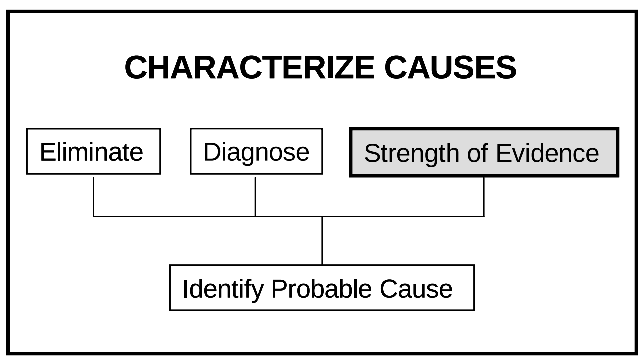
A state-wide study by Yoder and Rankin (1995b) indirectly examined the plausibility of specific community changes associated with nine types of sources, including waste water treatment plants, industrial point sources, conventional municipal sources, combined sewer overflows, channelization, and agricultural non-point sources. They found that deformities, erosions, lesions, tumors and anomalies (DELTA) in fish were associated with industrial discharges (Yoder and Rankin 1995b) and nutrient enrichment (Rankin et al. 1999). In the Little Scioto, the greatest % DELTA values are associated with the greatest nutrient concentrations. Among macroinvertebrates, the loss of Tanytarsini midges and the increase of *Cricotopus sp*. are both associated with industrial discharges (Yoder and Rankin 1995b). In the Little Scioto, the disappearance of Tanytarsini midges and an increase in *Cricotopus* are associated with Impairment C.

Table 7-8. Comparison of the reported concentration of water quality parameters (mg/L) with exceedances.

Sediment Parameter Criteria mg/L	**RM 7.9 [RM 7.1]**	**RM 6.5**	**RM 5.8 [RM 6.2]**
Ammonia[a] 0.57 mg/L at pH 8.5 1.27 mg/L at pH 8.0	(<0.05) [0.11, <0.05]	(0.1)	(1.2) [0.35, 0.69]
Dissolved Oxygen[b] 3.0 mg/L for MWH	{4.6-2.8}* (7.9, 5.7)	{7.2-1.9}* (NA)	{8.3, 4.2} (8.23, 4.21)
Nitrate-nitrite[c] 1.6 mg/L	(1.4) [0.73, <0.1]	(0.8)	(8.1)* [0.33, 2.37]*
Total phosphorus[d] 0.28 mg/L	(0.07) [0.36*, 0.13]	(0.09)	{1.65}* (2.17)* [1.9, 1.21]*

No Entry = No data for that year. { } = 1987, ( ) = 1992, [ ] = 1998.
[a] USEPA (1998b) recommended ammonia criterion
[b] OEPA (1994) dissolved oxygen criterion
[c] Rankin et al. (1999) proposed nitrate-nitrite criterion
[d] Rankin et al. (1999) proposed total phosphorus criterion
* Exceedance of criterion
Dissolved oxygen values are maximum and minimum. Ammonia, nitrate-nitrite, total phosphorus measured in August and October, 1998.

7.9 Characterize Causes: Strength of Evidence



Strength of evidence analysis uses all of the evidence generated in the analysis phase to examine the credibility of each remaining candidate cause. The causal considerations for the strength of evidence analyses used three types of evidence: case-specific evidence, evidence from other situations or biological knowledge, and evidence based on multiple lines of evidence (Section 4.3.3). All the evidence was evaluated for consistency or coherence with the hypothesized causes.

The results of the strength of evidence analysis are presented in Tables 7-9 to 7-11. Following the strength of evidence analysis, the candidate causes are characterized (Table 7-12). This involves describing the causal evidence and identifying the probable cause.

Table 7-9. Strength of evidence analysis for the three candidate causes of Impairment A, RM 7.9.

Causal Considera-tion	Evidence	Score	Evidence	Score	Evidence	Score
Case-Specific Considerations						
	Habitat Alteration		Metals Contamination		Nutrient Enrichment	
Co-occurrence	Compatible: At and below RM 7.9, the habitat of the Little Scioto is altered as a result of channelization. The degree of habitat alteration remains about the same to the mouth of the river. The upstream reference is not channelized and habitat is good.	+	Compatible: All sediment metal concentrations were slightly higher at RM 7.9 compared to upstream.	+	Compatible: N was elevated by 0.2mg/L in 1992 compared to upstream. P is the same or decreases compared to upstream.	+
Temporality	No evidence	NE	No evidence	NE	No evidence	NE
Consistency of Association	No evidence	NE	No evidence	NE	No evidence	NE
Biological Gradient	Not applicable: Other downstream candidate causes interfere with this consideration.	NA	Not applicable: Other downstream candidate causes interfere with this consideration.	NA	Not applicable: Other downstream candidate causes interfere with this consideration.	NA
Complete Exposure Pathway	Evidence for all steps: The fish and invertebrates inhabit the channelized reach where the habitat is altered. Channel was deepened. DO was depressed. Substrate was embedded.	++	Incomplete evidence: No internal concentrations of metals were measured. Metals were present in sediment and exposure could occur from ingestion or by respiration of epibenthic water or sediment particles or through the food chain.	+	Incomplete evidence: Fish and invertebrates inhabit stream where nutrients are elevated. Concentrations of algae or chlorophyll *a* were not measured.	+
Experiment	No evidence.	NE	No evidence.	NE	No evidence.	NE

NE = no evidence; NA = not applicable/not available

Table 7-9 (continued). Strength of evidence analysis for the three candidate causes of Impairment A, RM 7.9.

Causal Considera-tion	Evidence	Score	Evidence	Score	Evidence	Score
Considerations Based on Other Situations or Biological Knowledge						
	Habitat Alteration		Metals Contamination		Nutrient Enrichment	
Plausibility: Mechanism	Increased Relative Weight: Plausible: Artificially deepened channel allows larger sized fish to survive.	+	Increased Relative Weight: Implausible: No known mechanism for metals. Metals usually cause a decrease in the relative weight of fish (Eisler 2000b).	-	Increased Relative Weight: Implausible: N is a nutrient for algal growth. Greater production of algae could provide additional food, increasing fish growth. However, the mechanism is implausible because N is generally not limiting (Allan 1995).	-
	Increased DELTA: Not known: No obvious mechanism other than stress.	0	Increased DELTA: Implausible: Metals do not cause fin erosion and lesions (Eisler 2000b).	-	Increased DELTA: Plausible: Nutrients are believed to create conditions that favor opportunistic pathogens and fungi that cause lesions, fin erosion and interfere with wound healing.	+
	Loss of species: Plausible: Embedded sediments remove forage, reproductive, and cover habitats for benthic fish including darters and benthic invertebrates including mayflies. Low DO is not tolerated by many species (Karr and Schlosser 1977, Yount and Niemi 1990, Rankin 1995).	+	Loss of species: Plausible: Metals are known to cause lethal and sub-lethal effects to invertebrates and fish that can extirpate species from a site (Eisler 2000b). Metals usually cause a decrease in the relative weight of fish (Eisler 2000b).	+	Loss of species: Plausible: Switching to an autochthonous energy source could alter species survival and community composition of fish and invertebrates.	+
Plausibility: Stressor-Response	Increased Relative Weight: No evidence.	NE	Increased Relative Weight: Not applicable: Implausible mechanism.	NA	Increased Relative Weight: Not applicable: Implausible mechanism.	NA
	Increased DELTA: No evidence.	NE	Increased DELTA: Not applicable: implausible mechanism.	NA	Increased DELTA: Inconcordant: magnitude of nutrient change too small to cause effect.	-
	Loss of species: No evidence. No quantitative evidence. Habitat alteration associated with channelization is generally believed to be an all or none situation affected by it's spatial extent and severity.	NE	Loss of species: Inconcordant. No metals exceeded *Hyalella azteca* PEL values in 1987, 1992 or 1998. The TEL value for arsenic was exceeded only in 1992. Metals cumulative toxic units exceeded PEL in 1992, but only by 0.1 units (USEPA 1996b).	-	Loss of species: Inconcordant. The magnitude of nutrient change was too small to account for the dramatic shifts in invertebrate and fish metrics. Proposed nitrogen criterion for Ohio was not exceeded (Rankin et al. 1999).	-

NE = no evidence; NA = not applicable/not available

Table 7-9 (continued). Strength of evidence analysis for the three candidate causes of Impairment A, RM 7.9.

Causal Considera-tion	Evidence	Score	Evidence	Score	Evidence	Score
Considerations Based on Other Situations or Biological Knowledge (cont'd)						
	Habitat Alteration		Metals Contamination		Nutrient Enrichment	
Consistency of Association	Increased Relative Weight: In most places.	++	Increased Relative Weight: Many exceptions.	-	Increased Relative Weight: No evidence.	NE
	Increased DELTA: In most places.	++	Increased DELTA: Many exceptions.	-	Increased DELTA: Many exceptions. At many sites in Ohio, DELTA was not increased by these levels of N (Rankin et al. 1999).	-
	Loss of species: In most places. Moderate increase in DELTA and loss of species are commonly associated with habitat alteration associated with channelization (Yoder and Rankin 1995b). Increased Relative Weight is also commonly increased with deepened channels (Personal Observation). Agricultural areas with channelization having similar stressors showed decreases in IBI and ICI component metrics (Edwards et al. 1984, Sheilds et al. 1998).	++	Loss of species: Many exceptions. At other sites in Ohio with similar metals concentrations, Relative Weight and DELTA were not increased and species were abundant. Personal observation of Ohio database.	-	Loss of species: Many exceptions. At many sites in Ohio, IBI and ICI scores were high at these levels of N (Rankin et al. 1999). High IBI and ICI cannot be achieved when many species are lost.	-

NE = no evidence; NA = not applicable/not available

Table 7-9 (continued). Strength of evidence analysis for the three candidate causes of Impairment A, RM 7.9.

Causal Considera-tion	Evidence	Score	Evidence	Score	Evidence	Score
Considerations Based on Other Situations or Biological Knowledge (cont'd)						
	Habitat Alteration		Metals Contamination		Nutrient Enrichment	
Specificity of Cause	Increased Relative Weight: One of a few: Deep channels or pools required for larger fish. Relative weight of fish is significantly correlated with drainage area, a surrogate for channel depth (Norton 1999).	++	Increased Relative Weight: Not applicable: Implausible mechanism.	NA	Increased Relative Weight: Not applicable: Implausible mechanism.	NA
	Increased DELTA: One of many.	0	Increased DELTA: Not applicable: Implausible mechanism.	NA	Increased DELTA: One of many.	0
	Loss of species: One of many.	0	Loss of species: One of many.	0	Loss of species: One of many.	0
Analogy	Not applicable	NA	Not applicable	NA	Not applicable	NA
Experiment	Increased Relative Weight: No evidence	NE	Increased Relative Weight: No evidence	NE	Increased Relative Weight: No evidence	NE
	Increased DELTA: No evidence	NE	Increased DELTA: No evidence	NE	Increased DELTA: No evidence	NE
	Loss of species: Concordant: Artificial riffle and pools improved invertebrate assemblage in the channelized Olentangy River (Edwards et al. 1984), and fish in Mississippi River (Sheilds et al. 1998).	+++				
Predictive Performance	No evidence	NE	No evidence	NE	No evidence	NE
Considerations from Multiple Lines of Evidence						
	Habitat Alteration		Metals Contamination		Nutrient Enrichment	
Consistency of Evidence	Increased Relative Weight: All consistent.	+++	Increased Relative Weight: Inconsistent: Implausible mechanism.	---	Increased Relative Weight: Inconsistent: Magnitude of change inconsistent with magnitude of effect.	---
	Increased DELTA: All consistent.	+++	Increased DELTA: Inconsistent: Implausible mechanism.	---	Increased DELTA: Inconsistent: Magnitude of change inconsistent with magnitude of effect.	----
	Loss of species: All consistent.	+++	Loss of species: Inconsistent - Although metals are present, the concentrations are unlikely to cause species extirpation.	---	Loss of species: Inconsistent: Magnitude of change inconsistent with magnitude of effect.	---
Coherence of Evidence	Increased Relative Weight, Increased DELTA, Loss of species: None.	0	Increased Relative Weight, Increased DELTA, Loss of species: None.	0	Increased Relative Weight, Increased DELTA, Loss of species: None.	0

NE = no evidence; NA = not applicable/not available

Table 7-10. Strength of evidence analysis for the five candidate causes of Impairment B, RM 6.5.

Causal Consideration	Evidence	Score	Evidence	Score
Case-Specific Considerations				
	PAH contamination		Metals Contamination	
Co-occurrence	Compatible: Sediment PAH concentrations were several orders of magnitude greater at RM 6.5 than upstream (Table 13).	+	Compatible: Lead, chromium, copper and mercury concentrations in sediment were two to ten times greater at RM 6.5 than upstream. Cadmium and zinc were also greater, but to a lesser degree.	+
Temporality	No evidence	NE	No evidence	NE
Consistency of Association	No evidence: only one location.	NE	No evidence: only one location.	NE
Biological Gradient	Not Applicable: Other candidate causes downstream interfere with this consideration.	NA	Not Applicable: Other candidate causes downstream interfere with this consideration.	NA
Complete Exposure Pathway	Actual evidence for all steps: PAHs were present in the sediment, and bottom-feeding fish and benthic invertebrates are typically exposed to sediment contaminants. Both BAP and NAPH metabolites were found in fish. EROD, a detoxifying enzyme known to be induced by PAH, was elevated.	+ +	Actual evidence for all steps: Metals were present in sediment and exposure could occur from ingestion or by respiration of epibenthic water of sediment particles or through the food chain. Zinc and lead were detected in fish tissues.	+ +
Experiment	No evidence	NE	No evidence	NE

NE = no evidence; NA = not applicable/not available

Table 7-10 (continued). Strength of evidence analysis for the five candidate causes of Impairment B, RM 6.5.

Causal Consideration	Evidence	Score	Evidence	Score
Considerations Based on Other Situations or Biological Knowledge				
	PAH contamination		Metals Contamination	
Plausibility: Mechanism	Decreased relative weight: Plausible: PAHs are known to reduce growth. Toxic compounds can shorten life span resulting in smaller fish (Eisler 2000a).	+	Decreased relative weight: Plausible: Metals are known to reduce growth. Toxic compounds can shorten life span resulting in smaller fish (Eisler 2000b).	+
	Increased DELTA: Plausible: PAHs are known to cause eroded barbels, fin erosion, lesions and internal and external tumors (Eisler 2000a).	+	Increased DELTA: Implausible: Metals do not cause fin erosion and lesions	-
	Decreased species: Plausible: PAHs are known to be toxic and cause reproductive impairments which could extirpate species (Eisler 2000a).	+	Decreased species: Plausible: Metals are known to cause lethal and sub-lethal effects to invertebrates and fish that can extirpate species from a site (Eisler 2000b).	+
Plausibility: Stressor-Response	Decreased relative weight: Concordant: Toxic levels are consistent with decreased fish growth.	+	Decreased relative weight: Ambiguous. Toxic levels are consistent with decreased fish growth (Eisler 2000b).	0
	Increased DELTA: Quantitatively consistent: PAHs are at levels that cause tumors and other DELTA.	+++	Increased DELTA: Not applicable: mechanism is implausible.	NA
	Decreased species: Quantitatively consistent: The *Hyalella azteca* PEL's were exceeded for all PAHs. The cumulative PAH toxic units ranged between 339 to 18,820 times the PEL value (USEPA 1996b).	+++	Decreased species: Quantitatively consistent. Lead exceeded *Hyalella azteca* PEL values in 1988-1991 and 1992 and chromium in 1992. The cumulative toxic units values for all metals range from 1.6 to 5.1 (USEPA 1996b).	+++
Consistency of Association	Decreased relative weight: In most places: Decreased relative weight is associated with complex toxic exposures (Yoder and Rankin 1995b).	++	Decreased relative weight: In most places: Decreased relative weight is associated with complex toxic exposures (Yoder and Rankin 1995).	++
	Increased DELTA: Invariant: Tumors and other DELTA are associated with fish exposed to high concentrations of PAH in fresh and marine waters (Albers 1995).	+++	Increased DELTA: Not applicable.	NA
	Decreased species: Invariant: At more than 25 locations associated with PAH contamination that exceeded exposure criteria in Ohio, IBI and ICI scores were below 30 (Cormier et al. 2000a). IBI and ICI are known to be depressed even when habitat quality is high (Cormier et al. 2000b, OEPA 1992a). IBI and ICI scores of less than 30 only occur when some species are extirpated.	+++	Decreased species: In most places: Hickey and Clements (1998) reviewed changes in invertebrate community associated with metals in water column.	++
Specificity of Cause	Decreased relative weight: One of many.	0	Decreased relative weight: One of many.	0
	Increased DELTA: One of many. PAHs are known to cause external lesions seen at Impairment B.	0	Increased DELTA: Not applicable.	NA
	Decreased species: One of many.	0	Decreased species: One of many.	0

NE = no evidence; NA = not applicable/not available

Table 7-10 (continued). Strength of evidence analysis for the five candidate causes of Impairment B, RM 6.5.

Causal Consideration	Evidence	Score	Evidence	Score
Considerations Based on Other Situations or Biological Knowledge (cont'd)				
	PAH contamination		Metals Contamination	
Analogy	Not applicable	NA	Not applicable	NA
Experiment	Decreased relative weight: Concordant: Following dredging in the Black River, Ohio, the age structure of the brown bullheads increased (Baumann and Harshbarger 1995).	+++	No evidence: No references sought.	NE
	Increased DELTA: Concordant: In the Black River Ohio, removal of PAHs by dredging resulted in lower levels of DELTA (Baumann and Harshbarger 1995) and PAH bile metabolites (Lin et al. submitted).	+++	No evidence: No references sought.	
	Decreased species: Concordant: Following dredging the composition of species at this site also changed (Baumann, pers. comm.).	+++	No evidence: No references sought.	
Predictive Performance	No evidence	NE	No evidence	NE
Considerations from Multiple Lines of Evidence				
	PAH contamination		Metals Contamination	
Consistency of Evidence	Decreased relative weight: All consistent.	+++	Decreased relative weight: All consistent.	+++
	Increased DELTA: All consistent.	+++	Increased DELTA: Multiple inconsistencies.	---
	Decreased species: All consistent.	+++	Decreased species: All consistent.	+++
Coherence of Evidence			Increased DELTA: No known explanation.	0

NE = no evidence; NA = not applicable/not available

Table 7-10 (continued). Strength of evidence analysis for the five candidate causes of Impairment B, RM 6.5.

Causal Consideration	Evidence	Score	Evidence	Score	Evidence	Score
Case-Specific Considerations						
	Ammonia Toxicity		Low Dissolved oxygen/High BOD		Nutrient Enrichment	
Co-occurrence	Compatible: Ammonia concentration was doubled relative to Impairment A.	+	Compatible: In 1992, BOD was double the upstream value and the lowest DO levels measured were 0.9 mg/L less than upstream.	+	Compatible: Compared to RM 7.9, P was elevated by 0.02 mg/L. N was less.	+
Temporality	No evidence	NE	No evidence	NE	No evidence	NE
Consistency of Association	No evidence: Only one location.	NE	No evidence: Only one location.	NE	No evidence: Only one location.	NE
Biological Gradient	Not applicable: Other downstream candidate causes interfere with this consideration.	NA	Not applicable: Other downstream candidate causes interfere with this consideration.	NA	Not applicable: Other downstream candidate causes interfere with this consideration.	NA
Complete Exposure Pathway	Evidence for all steps: Fish and invertebrates inhabited stream where ammonia was present.	+ +	Evidence for all steps: Fish and invertebrates inhabited stream where conditions of low DO and high BOD occurred.	+ +	Evidence for all steps: Fish and invertebrates inhabit stream where P was elevated.	+ +
Experiment	No evidence	NE	No evidence	NE	No evidence	NE

NE = no evidence; NA = not applicable/not available

Table 7-10 (continued). Strength of evidence analysis for the five candidate causes of Impairment B, RM 6.5.

Causal Consideration	Evidence	Score	Evidence	Score	Evidence	Score
Considerations Based on Other Situations or Biological Knowledge						
	Ammonia Toxicity		Low Dissolved oxygen/High BOD		Nutrient Enrichment	
Plausibility: Mechanism	Decreased relative weight: Plausible: Ammonia toxicity could reduce growth and survival. Low survival could alter the age structure resulting in smaller, younger fish.	+	Decreased relative weight: Plausible: Stress could reduce growth and survival. Low survival could alter the age structure resulting in more smaller, younger fish.	+	Decreased relative weight: Implausible: Increased nutrients are usually associated with increased algal growth that augment the energy available for growth.	–
	Increased DELTA: Plausible: Ammonia has been associated with anomalies (Dyer, pers. comm.).	+	Increased DELTA: Not known: No known mechanism.	0	Increased DELTA: Plausible: Nutrients are believed to create conditions that favor opportunistic pathogens and fungi that cause lesions, fin erosion, and interfere with wound healing (Rankin et al. 1999).	+
	Decreased species: Plausible: Ammonia is known to be toxic to fish and invertebrates (USEPA 1998b).	+	Decreased species: Plausible: Low DO can kill fish and invertebrates (Allan 1995).	+	Loss of species: Plausible: Switching to an autochthonous energy source could alter species survival and community composition for fish and invertebrates (Allan 1995).	+
Plausibility: Stressor-Response	Decreased relative weight: No evidence.	NE	Decreased relative weight: No evidence.	NE	Decreased relative weight: Inconcordant.	-
	Increased DELTA: No evidence.	NE	Increased DELTA: Not applicable.	NA	Increased DELTA: Inconcordant.	-
	Decreased species: Inconcordant: The ammonia concentrations were not great enough to cause the dramatic effects seen at Impairment B. Ammonia criteria were not exceeded. (USEPA 1998b).	-	Decreased species: DO levels are below Ohio criteria for MWH (OEPA 1992b).	+	Decreased species: Inconcordant: The magnitude of P change was not great enough to cause dramatic effects seen at Impairment B. Proposed P criterion was not exceeded (Rankin et al. 1999).	-

NE = no evidence; NA = not applicable/not available

Table 7-10 (continued). Strength of evidence analysis for the five candidate causes of Impairment B, RM 6.5.

Causal Consideration	Evidence	Score	Evidence	Score	Evidence	Score
Considerations Based on Other Situations or Biological Knowledge (cont'd)						
	Ammonia Toxicity		Low Dissolved oxygen/High BOD		Nutrient Enrichment	
Consistency of Association	No evidence	NE	No evidence.	NE	Decreased relative weight: Many exceptions.	-
					Increased DELTA: Many exceptions: DELTA are associated with increased P at many sites in Ohio, but at a higher concentration of P (Rankin et al. 1999).	-
					Decreased species: Many exceptions: Reduced species are associated with many sites in Ohio increased P, but at a higher concentration (Rankin et al. 1999).	-
Specificity of Cause	Decreased relative weight: One of many.	0	Decreased relative weight: One of many	0	Decreased relative weight: Not applicable	NA
	Increased DELTA: One of many.	0	Increased DELTA: Not applicable.	NA	Increased DELTA: One of many.	0
	Decreased species: One of many.	0	Decreased species: One of many.	0	Decreased species: One of many.	0
Analogy	Not applicable	NA	Not applicable	NA	Not applicable	NA
Experiment	No evidence: No reference sought.	NE	No evidence: No reference sought.	NE	No evidence: No references sought.	NE
Predictive Performance	No evidence	NE	No evidence	NE	No evidence	NE

NE = no evidence; NA = not applicable/not available

Table 7-10 (continued). Strength of evidence analysis for the five candidate causes of Impairment B, RM 6.5.

Causal Consideration	Evidence	Score	Evidence	Score	Evidence	Score
Considerations from Multiple Lines of Evidence						
	Ammonia Toxicity		Low Dissolved oxygen/High BOD		Nutrient Enrichment	
Consistency of Evidence	Decreased relative weight: All consistent.	+++	Decreased relative weight: Most consistent.	+	Decreased relative weight: Many inconsistencies.	_ _ _
	Increased DELTA: All consistent.	+++	Increased DELTA: Many inconsistencies: No known mechanism.	_ _ _	Increased DELTA: Many inconsistencies: Magnitude of change inconsistent with magnitude of effect.	_ _ _
	Decreased species: Inconsistent: Magnitude of change inconsistent with magnitude of effect.	_ _ _	Decreased species: Most consistent.	+	Decreased species: Many inconsistencies: Magnitude of change inconsistent with magnitude of effect.	_ _ _
Coherence of Evidence	Decreased species: No known explanation.	0	Increased DELTA: No known explanation.	0	Decreased relative weight, Increased DELTA, Decreased species: No known explanation.	0

NE = no evidence; NA = not applicable/not available

Table 7-11. Strength of evidence analysis for the three candidate causes of Impairment C, RM 5.7.

Causal Consideration	Evidence	Score	Evidence	Score	Evidence	Score
Case-Specific Considerations						
	Metals Contamination		Ammonia Toxicity		Nutrient Enrichment	
Co-occurrence	Uncertain: There were only slight changes in metal concentrations in sediment at RM 5.7 compared to RM 6.5. Only copper and zinc increased slightly and possibly cadmium. All others declined.	0	Compatible: Ammonia concentrations were 10X or greater than at RM 6.5. from RM 5.7 to RM 2.7	+	Compatible: Total phosphorus and nitrogen concentrations are elevated at RM 5.7 through 2.7. P values are more than 24X greater than at RM 6.5 and more than 10X greater for nitrogen than upstream.	+
Temporality	No evidence	NE	No evidence	NE	No evidence	NE
Consistency of Association	Similar patterns of fish and invertebrate communities are seen at RM 5.7, 4.4 and 2.7	+	Similar patterns of fish and invertebrate communities are seen at RM 5.7, 4.4 and 2.7.	+	Similar patterns of fish communities are seen at RM 5.7, 4.4 and 2.7.	+
Biological Gradient	Increased DELTA: Strong and monotonic: From RM 5.7 to RM 0.4, copper and mercury are strongly correlated with % DELTA.	++	Increased DELTA: None: No correlation of ammonia with % DELTA.	-	Increased DELTA: Strong and monotonic: % DELTA was moderately correlated with BOD, N and P.	++
	Decreased Tanytarsini: Strong and monotonic: The decline in % tanytarsini was also strongly correlated with copper and mercury.	++	Decreased Tanytarsini: None: No correlation of ammonia with the decline in % Tanytarsini.	-	Decreased Tanytarsini: Strong and monotonic: BOD, nitrate-nitrite and phosphorus were all strongly correlated with decline in % Tanytarsini midges and the ICI.	++

NE = no evidence; NA = not applicable/not available

Table 7-11 (continued). Strength of evidence analysis for the three candidate causes of Impairment C, RM 5.8

Causal Consideration	Evidence	Score	Evidence	Score	Evidence	Score
Case-Specific Considerations (cont'd)						
	Metals Contamination		Ammonia Toxicity		Nutrient Enrichment	
Complete Exposure Pathway	Incomplete evidence: Lead and zinc were detected in water samples (OEPA 1992a). In sediment, many metals were detected. No internal concentrations of metals were measured. Water hardness may have reduced metal availability.	+	Evidence for all steps: Ammonia levels measured in water column, so exposure possible for fish and invertebrates. Ammonia is directly discharged into streams by point sources. Temperature and pH conditions are favorable for forming unionized ammonia, the toxic form of ammonia. Conditions are favorable for conversion of nitrites to ammonia (low DO).	+ +	Incomplete evidence: Nutrient and phosphorus concentrations were measured in water column, and would be available for algal, fungal and bacterial growth. Neither algal nor chlorophyll *a* concentrations, the direct effect of nutrient enrichment, nor bacterial concentrations were not measured.	+
Experiment	No evidence.	NE	No evidence.	NE	No evidence.	NE
Considerations Based on Other Situations or Biological Knowledge						
Plausibility: Mechanism	Increased DELTA: Implausible: Metals do not cause fin erosion and lesions (Eisler 2000b).	-	Increased DELTA: Plausible: Ammonia has been associated with DELTA (Dyer, pers. comm.).	+	Increased DELTA: Plausible: Nutrients are believed to create conditions that favor opportunistic pathogens and fungi that cause lesions, fin erosion and interfere with wound healing (Rankin et al. 1999).	+
	Decreased Tanytarsini: Plausible: Metals are known to cause lethal and sub-lethal effects to invertebrates that can extirpate species from a site. In a literature review, lead and copper were associated with mortality and other metals with mortality, reproduction, growth and behavior changes (Eisler 2000b).	+	Decreased Tanytarsini: Plausible: Ammonia is toxic to benthic macroinvertebrates (USEPA 1998b).	+	Decreased Tanytarsini: Increased nutrients are known to change community structure primarily by changing the food source (Allan 1995).	+

NE = no evidence; NA = not applicable/not available

Table 7-11 (continued). Strength of evidence analysis for the three candidate causes of Impairment C, RM 5.8.

Causal Consideration	Evidence	Score	Evidence	Score	Evidence	Score
Considerations Based on Other Situations or Biological Knowledge (cont'd)						
	Metals Contamination		Ammonia Toxicity		Nutrient Enrichment	
Plausibility: Stressor-Response	Increased DELTA: Not applicable. Mechanism not plausible.	NA	Increased DELTA: Concordant	+	Increased DELTA: Quantitatively consistent: %DELTA consistent with associations of P concentrations found in streams throughout Ohio (Rankin et al, 1999)	+++
	Decreased Tanytarsini: Ambiguous: The cumulative toxic units exceed PEL by 1.5 to 2.8 times in 1988/91 and 1992, respectively. The cumulative toxic units for PEL decreased compared to upstream in 1988/91 and 1992. In 1998, cumulative PEL was 3.5 times greater than at Impairment B, but this occurred after the impairment had already occurred (USEPA 1996b).	0	Decreased Tanytarsini: Quantitatively consistent: Ammonia concentrations are in a plausible range to cause toxic effects especially on warm, sunny days. Conservatively, ammonia was two times the USEPA chronic criteria (USEPA 1996b).	+++	Decreased Tanytarsini: Concordant. Nutrient criteria are proposed for Ohio and were exceeded at RM 5.7 through RM 0.4 for both nitrate-nitrite and phosphorus. At RM 5.7, nitrogen concentration was five times the proposed criterion value. P concentration was more than seven times the proposed phosphorus criterion (Rankin et al. 1999).	+
Consistency of Association	Increased DELTA: Many exceptions. Ohio EPA database.	-	Increased DELTA: In most places (Rankin et al. 1999).	++	Increased DELTA: In most places (Rankin et al. 1999).	+ +
	Decreased Tanytarsini: No evidence.	NE	Decreased Tanytarsini: No evidence.	NE	Decreased Tanytarsini: No evidence.	NE
Specificity of Cause and Effect	Increased DELTA: Not applicable.	NA	Increased DELTA: One of a few.	++	Increased DELTA: One of a few.	++
	Decreased Tanytarsini: One of many.	0	Decreased Tanytarsini: One of many.	++	Decreased Tanytarsini: One of many.	++
Analogy	Not applicable	NA	Not applicable	NA	Not applicable	NA
Experiment	No evidence	NE	No evidence	NE	No evidence	NE
Predictive Performance	No evidence	NE	No evidence	NE	No evidence	NE

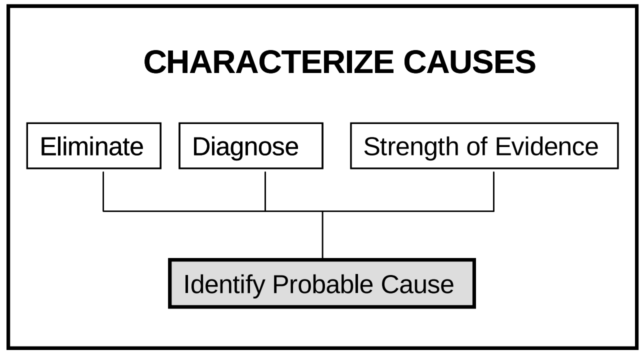
NE = no evidence; NA = not applicable/not available

Table 7-11 (continued). Strength of evidence analysis for the three candidate causes of Impairment C, RM 5.8.

Causal Consideration	Evidence	Score	Evidence	Score	Evidence	Score
Considerations from Multiple Lines of Evidence						
	Metals Contamination		Ammonia Toxicity		Nutrient Enrichment	
Consistency of Evidence	Increased DELTA: Multiple inconsistencies.	---	Increased DELTA: Most consistent.	+	Increased DELTA: All consistent.	+++
	Decreased Tanytarsini: Most consistent. Although metals are toxic the magnitude and type of effect do not seem to indicate that metals caused either the increase % DELTA or shifts in invertebrate metrics. However, mercury and copper are both significantly correlated with % DELTA and % tanytarsini.	0	Decreased Tanytarsini: Most consistent. Ammonia may have toxic effects, but % DELTA not likely to be caused by ammonia. No biological correlation.	+	Decreased Tanytarsini: All consistent. Reasonable evidence to suspect that nitrogen and phosphorus are creating conditions that favor opportunistic pathogens. Proposed criteria values are exceeded and high % DELTA consistent with effects seen even in the absence of toxics. Shifts in invertebrate metrics more uncertain.	++
Coherence of Evidence	Increased DELTA: No known explanation.	0	Increased DELTA: Biological gradient based on few observations and may be confounded by other stressors downstream.	0		
		0	Decreased Tanytarsini: Biological gradient based on few observations and may be confounded by other stressors downstream.	0		

NE = no evidence; NA = not applicable/not available

7.10 Characterize Causes: Identify Probable Causes



Impairment A (RM 7.9). At RM 7.9, there is a decline in IBI and ICI that is characterized by an increase in the relative weight of fish and percent DELTA, a decreased number of fish and species of fish, and a decreased percentage of mayflies. Candidate Causes #2, PAH, #4, ammonia, and #5, low DO/BOD were eliminated (Tables 7-5 and 7-6). Candidate causes #1, habitat alteration, #3, metal contamination, and #6, nutrient enrichment, were evaluated in a strength of evidence analysis (Tables 7-9, 7-10 and 7-11). An artificially deepened channel was identified as the probable cause for an increase in the relative weight of fish. An embedded stream bed was identified as the probable cause for decreased numbers and species of fish and decreased percentage of mayflies. The stream bed may have been susceptible to becoming embedded due to a lower gradient than upstream. The probable cause for the low but measurable increase in percent DELTA remained uncertain. The strength of evidence analysis strongly supports this causal relationship. The quality of the data is high, and the consistency of the evidence is good.

Impairment B (RM 6.5). At RM 6.5, there is a further decline in the IBI and ICI. Specific impairments include an increase in % DELTA, a decrease in the relative weight and numbers of species of fish, and an additional decrease in percent mayflies. Habitat alteration was eliminated as a candidate cause (Tables 7-5 and 7-6). In the strength of evidence analysis a single probable cause, PAHs, was found to be sufficient to cause all of the specific impairments (Tables 7-10 and 7-12). Habitat alteration continued to impair the site but was not the cause of the increased DELTA, decreased relative weight, or the additional decline in the number of species. The strength of evidence analysis strongly supports this causal relationship. The quality of the data is high, and the consistency of the evidence is very good.

Impairment C (RM 5.7). At RM 5.7, there is a notable further increase in % DELTA and a decrease in % Tanytarsini. Altered habitat and PAH still cause impairments, but since the level of alteration remains about the same or decreases, these candidate causes were eliminated (Tables 7-5 and 7-6). In the strength of evidence analysis, nutrient enrichment, candidate cause #6, was identified as the probable cause for both impairments. Nevertheless, ammonia toxicity may still be important. We have moderate confidence in this characterization.

The causal characterization of the Little Scioto River could be strengthened by evidence from published literature that reports associations applying to plausible mechanism and stressor-response, consistency of association, specificity, and others. It was not the intent of this document to prepare an exhaustive list of appropriate evidence, but such a resource is certainly needed to make these types of evidence accessible for future characterizations. This case study does demonstrate the stressor identification process and the importance of clearly presenting the reasoning and evidence.

Table 7-12. Causal characterization.

Impairment A - RM 7.9	**Impairment B - RM 6.5**	**Impairment C - RM 5.7**
Probable Cause: Habitat Alteration	Probable Cause: PAH Contamination	Probable Cause: Nutrient Enrichment
Increased Relative Weight: Is probably caused by the artificial deepening of the channel that allows larger fish to live there. Increased DELTA: The percentage of DELTA is commonly associated with channelized streams, but the specific aspect of the channelization that increased DELTA is unknown. Loss of species: Many factors could contribute to the loss of fish and benthic invertebrate species; however, embedded substrates seem to be the most likely stressor since upstream locations had even lower DO levels and yet had a greater variety of fish and invertebrate species. Although metals are present, the likelihood of response at the these concentrations are low. Furthermore, the types of changes in the community, especially an increase in the relative weight of fish, is very unlikely with the candidate cause of metals. Although P levels are slightly higher, effects are not associated with these phosphorous concentration elsewhere and they do not exceed Ohio's proposed criteria values for effects. Candidate Causes #2, PAH, and #4, Ammonia, were eliminated because levels were the same or lower than upstream. Candidate Cause #5, Low DO /BOD , was also eliminated as an overall pathway; however, low DO associated with channelization may still play a roll especially in DELTA. Siltation and deepened channel are consistent with Impairment A. The magnitude of the alteration and clear difference from upstream location strongly support this cause.	A single cause is likely for the three manifestations of Impairment B: decreased relative weight, increased DELTA, and decreased species: The probable cause of Impairment B is toxic levels of PAH-contaminated sediments. All of the evidence support PAH contamination as the cause. There is a complete exposure pathway at the location and clear mechanism of action for each of the effects. The single most convincing piece of evidence is that the cumulative toxic units of PAH were more than 300 times the probable effects level. Metals are at sufficient concentrations to cause effects; however, they were sometimes at levels close to upstream levels and were less than 2% as toxic as the lowest cumulative toxic units of PAH. Metal concentrations are high enough that they should be considered a potentially masked cause. Candidate cause #5 is unlikely because even greater levels of BOD did not cause reduction of dissolved oxygen downstream. Candidate Causes #4, Ammonia, and #6, Nutrient Enrichment, are unlikely given that state criteria levels were met and the much stronger evidence for PAH. Habitat alteration continues to impair the site, but it is not the cause of the increased DELTA, decreased relative weight, or the additional decline in the number of species.	At Impairment C increased % DELTA and % Tanytarsini may have different causes. Increased DELTA in fish is probably caused by increased P and NO_x. Nutrients, especially P, have been associated with increased fin erosion and lesions but some uncertainty exists since P acts indirectly. Ammonia is slightly higher than at Impairment B and exceeded ammonia criteria values. Biological gradients were absent; however, this may have been a statistical artifact given the number of sites available to perform the analysis and potential interference from other stressors downstream. Metals are considered unlikely because surface lesions are only occasionally noted as effects from long term exposure and only some metal concentrations were slightly greater than at Impairment B. Metal concentrations are high enough that they should be considered a potentially masked cause. The probable cause of extirpation of Tanytarsini at Impairment C is more uncertain because less is known about the natural history and stressor response relationships of these benthic invertebrates. Candidate cause #6, nutrient enrichment, still seems to be the most likely cause since all of the strength of evidence considerations were consistent. PAH contamination and habitat alteration continue to impair the site, but they are not the cause of the increased % DELTA or extirpation of Tanytarsini. The causal characterization at Impairment C is less certain, but the strength of evidence favors cause #6, increased nutrients.

7.11 Discussion

An important, practical aspect of this study is that even though the primary cause was identified in each case, it is obvious that other causes are also present that would constrain the biological community if the dominant cause was removed. For instance, if PAHs could be independently removed from the river, metals might be high enough to impair the biological assemblage. Likewise, if metals were removed, habitat alteration would still affect the biological community and would lower IBI and ICI scores at Impairments B and C.

Another issue is the impact of habitat alteration and its influence on modifying the assimilative capacity of the river. In other words, if the physical habitat were improved, would the impacts of PAH contamination be lessened? At Impairment B, this is unlikely based on evidence from at least one river elsewhere that has very good physical habitat qualities, yet has an impoverished biological community replete with high levels of % DELTA due to high PAH concentrations (OEPA 1992b, Cormier et al. 2000b). The strength of evidence analysis can provide these insights for the next step in managing ecosystems, which is to find ways to identify and apportion the sources for the identified causes and then take action to restore and protect the resource.

At Impairment C, a physical habitat that included wetlands, riparian wetlands, and riparian cover might improve the assimilative capacity of the river by providing sinks for the nutrient and ammonia loadings. However, since PAH and metals contamination are still high at Impairment C, removal of nutrient loading alone would result in only a very small improvement in biological condition.

At Impairment B, nutrient enrichment was retained as a candidate cause, even though the increase in phosphorous was minute. Nutrient enrichment was an unlikely cause, but the reasons for it being improbable come from ecological knowledge from examples in other watersheds, not from evidence that permits elimination. The reason nutrient enrichment was retained was because it failed to meet the criteria for elimination. The strength of evidence is the proper way to show this evidence.

There are other uncertainties. Wet weather flow data was not available for review. Events, especially near the combined sewer overflow at RM 6.0, could be undetected sources of candidate causes. Downstream from Impairment C, persistent impairments may have other causes. For instance, BOD is elevated at RM 5.8; however, its effects are usually associated with a certain lag time that results in low DO.

The results from this particular causal analysis could have several practical applications. If it is determined that the river conditions must be improved due to state regulations, federal TMDL (total maximum daily load) rules, citizen action, or other reasons, one option is to remove or decrease all potential stressors identified in the causal analysis; that is, remove both channel modification as well as water and sediment contamination. However, there may be intermediate pathways that may be more cost effective. Factors that should be considered in choosing an option include the desired or expected level of improvement in river condition, and the usefulness of the river's resources versus the cost to restore the river. Another factor to consider is the mode of restoration. For instance, both PAH and metal remediation may require dredging of the contaminated sediments. Knowing which agents (PAH, metals, or a combination of the two) may satisfy our curiosity, but it may not change the management action or ecological outcome. However, it might be determined that knowing the cause is important for assigning the financial responsibility for clean-up. In the latter case, additional information may be needed, especially if restoration costs are high.

7.12 References

Albers, P. 1995. Petroleum and individual polycyclic aromatic hydrocarbons, Pages 330-355 *in* Hoffman, David J. et al, ed. *Handbook of Ecotoxicology*, CRC Press, Boca Raton, Florida.

Allan, J.D. 1995. *Stream ecology: structure and function of running waters*. Chapman and Hall Publishers, London.

Baumann, P.C., and Harshbarger, J.C. 1995. Decline in liver neoplasms in wild brown bullhead catfish after coking plant closes and environmental PAHs plummet. *Environ. Health Perspect.* 103:168-170.

Carpenter, S.R., N.F. Caraco, D.L. Correll, R.W. Howarth, A.N. Sharpley and V.H. Smith. 1988. Non-point pollution of surface waters with phosphorus and nitrogen. *Ecological Applications* 8(3):559-568.

Cormier, S.M., E.L.C. Lin, F.A. Fulk and B. Subramanian. 2000a. Estimation of exposure criteria values for biliary polycyclic aromatic hydrocarbon metabolite concentration in white suckers (*Catostomus commersoni*). *Environmental Toxicology and Chemistry* 19:1120-1126.

Cormier, S.M., E.L.C. Lin, M.R. Millward, M.K. Schubauer-Berigan, D. Williams, B. Subramanian, R. Sanders, B. Counts and D. Altfater. 2000b. Using regional exposure criteria and upstream reference data to characterize spatial and temporal exposures to chemical contaminants. *Environmental Toxicology and Chemistry* 19:1127-1135.

Dodds, K. and E.B. Welsh. 2000. Establishing nutrient criteria in streams. *The North American Benthological Society* 19:186-196.

Edwards, R., and J. Riepenhoff. 1998. State turns to feds for cleanup. *Columbus Dispatch*, April 28, 1998.

Edwards, A.C., H. Twist and G.A. Codd. 2000. Assessing the impact of terrestrially derived phosphorus on flowing water systems. *Journal of Environmental Quality*. 29:117-124.

Edwards, C.J., B.L. Griswold, R.A. Tubb, E.C. Weber, L.C. Woods. 1984. Mitigating effects of artificial riffles and pools on the fauna of a channelized warmwater stream. *North American Journal of Fisheries Management* 4:194-203

Eisler, R. 2000a. Polycyclic aromatic hydrocarbons. Pages 1343-1411 *in Handbook of Chemical Risk Assessment*. Vol. II. Lewis Publishers, Boca Raton, FL.

_______. 2000b. *Handbook of Chemical Risk Assessment*. Vol. I. Lewis Publishers, Boca Raton, FL.

Gmur, D.J., and U. Varanasi. 1982. Characterization of benzo[a]pyrene metabolites isolated from muscle, liver, and bile of a juvenile flatfish. *Carcinogenesis* 5:1397-1403.

Hickey, C.W. and W.H. Clements. 1998. Effects of heavy metals on benthic macroinvertebrate communities in New Zealand streams. *Environmental Toxicology and Chemistry* 17:2338-2346

Karr, J.R., and I.J. Schlosser. 1977. *Impact of nearstream vegetation and stream morphology on water quality and stream biota.* EPA-600-3-77-097. U.S. Environmental Protection Agency, Environmental Research Laboratory, Athens, GA.

Lin, E.L.C., S.M. Cormier, and J.A. Torsella. 1996. Fish biliary polycyclic aromatic hydrocarbon metabolites estimated by fixed-wavelength fluorescence: comparison with HPLC-fluorescent detection. *Ecotoxicol. Environ. Safety* 35:16-23.

Lin, E.L.C., T.W. Neiheisel, B. Subramanian, D.E. Williams, M.R. Millward, and S.M. Cormier. Historical monitoring of biomarkers of exposure of brown bullhead in the remediated Black River, Ohio and two other Lake Erie tributaries. Submitted to *Journal of Great Lakes Research*.

Long, E.R., L.J. Field, and D.D. MacDonald. 1998. Predicting toxicity in marine sediments with numerical sediment quality guidelines. *Envir. Toxicol. Chem.* 17:714-727.

Meyer, F.P. and L.A. Barclay. 1990. *Field manual for the investigation of fish kills.* Resource Pub. 177. U.S. Fish and Wildlife Service, Washington, D.C.

Miltner, R.J. and E.T. Rankin. 1998. Primary nutrients and the biotic integrity of rivers and streams. *Freshwater Biology* 40:145-158.

Norton, S.B. 1999. *Using biological monitoring data to distinguish among types of stress in streams of the Eastern Cornbelt Plains Ecoregion.* Ph.D. Dissertation, Georgetown University, Fairfax, VA.

Ohio Environmental Protection Agency (OEPA). 1988a. *Biological criteria for the protection of aquatic life: Vol. II. users manual for biological assessment of ohio surface waters.* Division of Water Quality Planning and Assessment, Ecological Assessment Section, Columbus, OH.

__________. 1988b. *Biological and water quality study of the Little Scioto River watershed*, Marion County, OH. OEPA Technical Report prepared by State of Ohio Environmental Protection Agency, Division of Surface Water, Columbus, OH.

__________. 1989a. *Addendum to: Biological criteria for the protection of aquatic life: Volume II. users manual for biological assessment of Ohio surface waters.* Division of Water Quality Planning and Assessment, Ecological Assessment Section, Columbus, OH.

__________. 1989b. *Biological criteria for the protection of aquatic life: Volume III. standardized field and laboratory methods for assessing fish and macroinvertebrate communities.* Division of Water Quality Planning and Assessment, Ecological Assessment Section, Columbus, OH.

__________. 1989c. *Manual of Ohio EPA surveillance methods and quality assurance practices.* Division of Environmental Services, Columbus, OH.

__________. 1992a. *Bottom sediment evaluation, Little Scioto River, Marion, Ohio.* Division of Water Quality Planning and Ecological Assessment Section, Columbus, OH.

__________. 1992b. *Biological and water quality study of the Ottawa River, Hog Creek, Little Hog Creek, and Pike Run.* OEPA Technical Report EAS/1992-9-7. Prepared by State of Ohio Environmental Protection Agency, Division of Surface Water, Columbus, OH.

__________. 1994. *Biological, sediment, and water quality study of the Little Scioto River, Marion, Ohio.* OEPA Technical Report EAS/1994-1-1. Division of Surface Water, Ecological Assessment Section, Columbus, OH.

Rankin, E.T. 1989. *The qualitative habitat evaluation index (QHEI): rationale, methods, and application*. State of Ohio Environmental Protection Agency, Division of Water Quality Planning and Assessment, Ecological Assessment Section, Columbus, OH.

__________. 1995. Habitat indices in water resource quality assessments. Pages 181-208 *in* W.S. Davis and T.P. Simon (editors). *Biological Assessment and Criteria.* Lewis Publishers, Boca Raton, Florida.

Rankin E., R. Miltner, C. Yoder and D. Mishne. 1999. *Association between nutrients, habitat, and the aquatic biota in Ohio rivers and streams.* Ohio EPA Technical bulletin MAS/1999-1. Ohio EPA, Columbus, OH.

Roubal, W.T., T.K. Lallier, and D.C. Malins. 1977. Accumulation and metabolism of C-14 labeled benzene, naphthalene, and anthracene by young coho salmon (*Oncorhynchus kisutch*) and starry flounder (*Platichthys stellatus). Arch. Environ. Contam. Toxicol.* 5: 513-529.

Russo, R.C. 1985. Ammonia, nitrate and nitrite. Pages 455-471 *in* G.M. Rand and S.A. Petrocelli (editors). *Fundamentals of Aquatic Toxicology*. McGraw Hill, Washington, D.C.

Sheilds, F.D. Jr., S.S. Knight and C.M. Cooper. 1998. Rehabilitation of aquatic habitats in warmwater streams damaged by channel incision in Mississippi. *Hydrobiologica* 382:63-86

Smith, V.H., G.D. Tilman and J.C. Nekola. 1999. Eutrophication: impacts of excess nutrient inputs on freshwater, marine and terrestrial ecosystems. *Environmental Pollution* 100:179-196.

Tarplee, W.H. Jr., D.E. Louder, and A.J. Weber. 1971. *Evaluation of the effects of channelization on fish populations in North Carolina's coastal plain streams.* North Carolina Wildlife Resources Commission, Raleigh, NC.

U.S. Environmental Protection Agency (USEPA). 1996b. Calculation and evaluation of sediment effect concentrations for the amphipod *Hyallela azteca* and the midge *Chironomus riparius*. Assessment and Remediation of Contaminated Sediments (ARCS) Program. Great Lakes National Program Office, Chicago, IL. EPA 905-R96-008.

__________. 1998b. *1998 update of ambient water quality criteria for ammonia*. Office of Water, Washington, DC. EPA 822-R-98-008.

Varanasi, U., J.E. Stein, M. Nishimoto, and T. Hom. 1983. Benzo[a]pyrene metabolites in liver, muscle, gonads, and bile of adult English sole (*Parophrys vetulus*). Pages 1221-1234 *in* Cooke, M. and A.J. Dennis, eds. *Polynuclear Aromatic Hydrocarbons: Formation, Metabolism, and Measurement*. Battelle, Columbus, OH, USA.

Yoder, C.O., and E.T. Rankin. 1995a. Biological criteria program development and implementation in Ohio. Pages 109-144 *in* W.S. Davis and T.P. Simon, eds. *Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making.* Lewis Publishers, Boca Raton, FL.

__________. 1995b. Biological response signatures and the area of degradation value: New tools for interpreting multi-metric data. Pages 236-286 *in Biological Assessment and Criteria: Tools for Water Resource Planning and Decisionmaking* Lewis Publishers, Boca Raton, FL..

Yount, J.D., and G.J. Niemi. 1990. Recovery of lotic communities and ecosystems from disturbance; A narrative review of case studies. *Environmental Management* 14:547-569.

Table 7-13. Fish metrics for the Little Scioto River 1987 and 1992.*

Response	River Mile							
	{9.2} (9.2)	(7.9)	{6.5} (6.5)	{6.0} (5.7)	(4.4)	(3.1)	{2.7} (2.7)	{0.1} (0.3)
Total No. of Species	{19.3} (22)	(13)	{13} (8.3)	{10} (10.7)	(7.7)	{3.3}	{6} (7.7)	{8.7} (9.7)
No. of Darter Species	{5 } (5.5)	(0)	{0} (0)	{0} (0)	(0)	{0}	{0} (0)	{0.3}
No. of Sunfish species	{3} (4)	(5)	{3} (2.3)	{1.3} (3.3)	(3)	{0.7}	{0.3} (2.7)	{1} (2.7)
No. of Sucker Species	{1.3} (2.5)	(3)	{1} (1.3)	{1.7} (1.7)	(1)	{0.7}	{1} (1.3)	{2} (2.7)
No. of Intolerant Species	{1} (1)	0	{0} (0)	{0} (0)	(0)	{0}	{0} (0)	{0}
Percent Tolerant Species	{35.45} (60.69)	(69.12)	{82.43} (85.12)	{94.28} (68.14)	(82.75)	{98.2}	{94.95} (70.85)	{63.41} (38.64)
Percent Omnivores	{33.28} (56.21)	(44.95)	{57.2} (56.72)	{72.47} (46.84)	(71.37)	{94.15}	{85.4} (51.77)	{62.72} (31.92)
Percent Insectivores	{53.41} (35.96)	(53.07)	{40.99} (39.77)	{16.73} (47.8)	(21.91)	{3.95}	{10.18} (42.51)	{32.74} (55.36)
Percent Pioneering Species	{35.49} (69.85)	(28.41)	{22.52} (26.39)	{21.94} (21.76)	(17.81)	{4.05}	{7.58} (23.31)	{5.33} (22.1)
No. of Individuals	{808.5} (1104.6)	(206)	{416.3} (335)	{237.33} (174)	(137)	{84.7}	{237.33 } (94)	{78} (75)
Percent Simple Lithophilic Species	{21.24} (12.8)	(18.19)	{2.7} (42.69)	{24.01} (26.2)	(31.45)	{5.88}	{9.14} (24.91)	{19.01} (28.08)
Percent DELTA	{0.13} (0.14)	(1.64)	{0.0} (9.98)	{16.46} (14.51)	(22.37)	{32.8}	{14.22} (10.99)	{16.19} (10.04)
Relative Weight	{4.021} (8.6)	(74.9)	{34.2} (38.7)	{29.773} (17.031)	(7.2)	{10.7}	{24.482 } (6.3)	{46.079} (21.1)
IBI	{332 (33)	(23)	{24} (19)	{14} (19)	(18)	{12}	{13} (19)	{14} (25)

Table 7-14. Macroinvertebrate metrics for the Little Scioto River 1987 and 1992.*

Response	River Mile							
	{9.2} (9.2)	(7.9)	{6.5} (6.5)	{5.8} (5.7)	(4.4)	{3.2}	{2.7} (2.1)	{0.4} (0.4)
Total Number of Macroinvertebrates	{773} (1464)	(1952)	{1116} (2815)	{207} (1600)	(1899)	{763}	{1779} (5242)	{645} (1151)
Total No. of Taxa collected at a Site, both Qualitative and Quantitative	{51} (47)	(38)	{38} (29)	{26} (32)	(27)	{28}	{28} (41)	{24} (37)
Total No. of Quantitative Taxa	{34} (36)	(30)	{25} (18)	{13} (18)	(20)	{14}	{13} (23)	{16} (26)
No. of Mayfly Taxa	{6} (7)	(2)	{3} (2)	{2} (2)	(1)	{1}	{0} (2)	{2} (3)
No. of Caddisfly Taxa	{2} (3)	(0)	{2} (0)	{0} (0)	(1)	{0}	{0} (3)	{0} (1)
No. of Dipteran Taxa	{19} (20)	(18)	{15} (12)	{7} (13)	(13)	{10}	{11} (14)	{12} (16)
No. of Qualitative EPT Taxa	{10} (8)	(1)	{2} (0)	{1} (1)	(0)	{4}	{2} (6)	{1} (2)
Percent Mayfly Taxa	{56.016} (58.811)	(16.393)	{20.251} (5.009)	{3.382} (2)	(0.263)	{1.573}	{0} (0.114)	{1.24} (3.215)
Percent Caddisfly Taxa	{4.657} (6.557)	(0)	{0.179} (0)	{0} (0)	(0.053)	{0}	{0} (0.267)	{0} (0.087)
Percent Tanytarsini Midges	{1.552} (3.347)	(3.381)	{4.48} (2.345)	{0.966} (0)	(0)	{3.67}	{0} (0.343)	{0} (2.085)
Percent Dipterans	{26.132} (32.445)	(74.795)	{55.018} (57.336)	{37.198} (91)	(74.829)	{95.937}	{23.834} (97.138)	{61.085} (91.659)
Percent Non-insects	{7.762} (1.639)	(5.43)	{20.43} (37.549)	{56.039} (7)	(21.959)	{0.524}	{75.998} (2.461)	{37.674} (4.344)
Percent Tolerant Organisms	{4.916} (8.607)	(15.061)	{37.993} (77.371)	{61.353} (67.75)	(54.766)	{20.315}	{89.545} (29.569)	{52.713} (59.34)
Percent *Cricotopus*	{0.388} (0.48)	(0)	{6.631} (0)	{0} (8)	(2.53)	{0}	{4.947} (0.301)	{4.961} (2.172)
ICI	{40} (38)	(16)	{22} (8)	{8} (6)	(10)	{8}	{4} (18)	{6} (18)

* { } = 1987; ( ) = 1992

Table 7-15. QHEI metrics for the Little Scioto River 1987 and 1992.*

Metric	**River Mile**								
	{9.2} (9.2)	(7.9)	{6.5} (6.5)	{6.0}	(5.7)	(4.4)	{3.1}	{2.7} (2.7)	{0.1} (0.3)
Substrate	{18} (16)	(5)	{1} (1)	{1}	(1)	(5)	{1}	{1} (5)	{1} (5)
Cover	{10} (14)	(10)	{9} (11)	{13}	(10)	(10)	{11}	{13} (11)	{12} (9)
Cover Types	{3} (6)	(4)	{2} (6)	{6}	(4)	(4)	{4}	{6} (6)	{5} (6)
Channel	{18} (17)	(10)	{6} (10)	{10}	(10)	(10)	{11}	{10.5} (10)	{10} (7)
Riparian	{9} (6)	(5.5)	{4} (4)	{4}	(6)	(6)	{5}	{6} (8)	{8} (5.5)
Pool	{8} (11)	(8)	{6} (8)	{8}	(9)	(6)	{6}	{8} (8)	{8} (8)
Riffle	{5} (6)	(0)	{0} (0)	{0}	(0)	(0)	{0}	{0} (0)	{0} (0)
Gradient	{6} (6)	(4)	{4} (4)	{4}	(4)	(2)	{2}	{2} (2)	{4} (4)
QHEI	{74} (76)	(42.5)	{30} (38.5)	{40}	(40)	(39)	{36}	{40.5} (42)	{43} (38.5)

* { } = 1987; ( ) = 1992

Table 7-16. Average concentrations of selected sediment organic compounds (mg/kg) in the Little Scioto River, Ohio, by river mile in 1987, 1991, 1992 and 1998.*

Compound	River Mile								
	[11.1]	/9.42\ (9.5) [9.21]	(7.9)	/7.15 \ [7.09]	{6.5} /6.6\ (6.5) [6.6]	/5.8\ (5.8) [6.2]	(4.4)	{2.7} /2.7\ (2.7) [2.65]	(0.4)
Acenaphthene	[0.59]ND	(ND) [0.7]ND	(ND)	[0.047]J	{14.8} (ND) [760]J	/150\ (5) [5]	(4.3)	{1.3} (ND) [0.930]J	(ND)
Anthracene	[0.59]ND	(ND) [0.70]ND	(ND)	[0.037]J	{66.8} (ND) [100]J	/360\ (27.1) [41]	(7.9)	{2.3} (ND) [3.7]	(3.3)
Benzo(a)anthracene	[0.072]J	(ND) [0.043]J	(ND)	/15\J [0.059]J	{44.7} /15\J (8.2)J [310]J	/185\ (16.5) [42]J	(6.9)	{4.3} (2)J [8.2]	(15.8)
Benzo(b)fluoranthene	[0.068]J	(ND) [0.052]J	(ND)	/25\ [0.051]J	{23.6} /20\J (18.1) [200]J	/215\ (16.8) [95]	(6.9)	{2.0} (1.6)J [12]	(13.8)
Benzo(k)fluoranthene	[0.058]J	(ND) [0.052]J	(ND)	[0.046]J	{213.2} (9.9)J [160]J	(12.87) [80]	(4.6)	{21.3} (ND) [10]	(10.5)

Table 7-16 (continued). Average concentrations of selected sediment organic compounds (mg/kg) in the Little Scioto River, Ohio, by river mile in 1987, 1991, 1992 and 1998.*

Compound	River Mile								
	[11.1]	/9.42\ (9.5) [9.21]	(7.9)	/7.15 \ [7.09]	{6.5} /6.6\ (6.5) [6.6]	/5.8\ (5.8) [6.2]	(4.4)	{2.7} /2.7\ (2.7) [2.65]	(0.4)
Benzo(ghi)perylene	[0.052]J	(ND) [0.044]J	(ND)	/10\J [0.030]J	{144.1} (49.5) [150]ND	/65\ (11.2) [19]	(4.9)	{16.5} (ND) [13]	(6.9)
Benzo(a)pyrene	[0.067]J	(ND) [0.053]J	(ND)	/10\J [0.043]J	{141.1} (14.8)J [210]J	/125\ (15.8) [14]	(7.2)	{11.4} (ND) [12]	(11.5)
Chrysene	[0.087]J	(ND) [0.065]J	(ND)	/15\J [0.081]J	{119.5} /15\J (16.5) [390]J	/305\ (20.8) [13]	(9.9)	{9.7} (1.6)J [13]	(ND)
Dibenzo(a,h)anthracene	[0.59]ND	(ND) [0.7]ND	(ND)	[0.56]N D	{33.3} (ND) [150]ND	(4.6) [16]ND	(ND)	{2.1} (ND) [3.7]	(3.3)
Fluoranthene	[0.19]J	(ND) [0.097]J	(ND)	/20\J [0.20]J	{78.4} /50\ (8.2)J [100]J	/550\ (37.6) [44]J	(13.5)	{6.3} (ND) [14]	(22.4)

Table 7-16 (continued). Average concentrations of selected sediment organic compounds (mg/kg) in the Little Scioto River, Ohio, by river mile in 1987, 1991, 1992 and 1998.*

Compound	River Mile								
	[11.1]	/9.42\ (9.5) [9.21]	(7.9)	/7.15 \ [7.09]	{6.5} /6.6\ (6.5) [6.6]	/5.8\ (5.8) [6.2]	(4.4)	{2.7} /2.7\ (2.7) [2.65]	(0.4)
Fluorene	[0.590]ND	(ND) [0.70]ND	(ND)	[0.059]J	{18.3} (ND) [830]J	/200\ (7.0) [20]	(4.0)	{1.2} (ND) [0.98]J	(ND)
Indeno(1,2,3-cd)pyrene	[0.045]J	(ND) [0.037]J	(ND)	/5\J [0.56]ND	{156.0} (13.2)J [150]ND	/60\ (14.5) [16]	(6.6)	{18.6} (ND) [10]	(10.5)
Naphthalene	[0.59]ND	(ND) [0.70]ND	(ND)	[0.56]ND	{22.9} (ND) [260]	/70\ (4.6) [18]J	(ND)	{1.6} (ND) [0.28]J	(ND)
Phenanthrene	[0.14]J	(ND) [0.053]J	(ND)	[0.11]J	{88.3} /40\J (ND) [230]	/470\ (24.1) [38]J	(12.9)	{2.0} (ND) [2.8]J	(2.6)J

* { } = 1987; / \ = 1991; ( ) = 1992; [ ] = 1998

Table 7-16 (Continued). Average concentrations of selected sediment organic compounds (mg/kg) in the Little Scioto River, Ohio, by river mile in 1987, 1991, 1992 and 1998.*

Compound	River Mile								
	[11.1]	/9.42\ (9.5) [9.21]	(7.9)	/7.15 \ [7.09]	{6.5} /6.6\ (6.5) [6.6]	/5.8\ (5.8) [6.2]	(4.4)	{2.7} /2.7\ (2.7) [2.65]	(0.4)
Pyrene	[0.2]J	(ND) [0.1]J	(ND)	/15\J [0.2]J	{67.5} /30\J (ND) [810]J	/405\ (23.8) [32]J	(10.2)	{5.2} (ND) [10]	(17.5)

{ } = 1987 data from OEPA 1988, sample depth unknown
/ \ = 1991 data from OEPA 1992a, sample depth unknown
( ) = 1992-93 data from OEPA 1994, sample from 1-6" except RM 7.9 sample from 8-12"
[ ] = 1998 data from OEPA unpublished, sample depth unknown
J is an estimated value that is above zero but below the practical quantitation limit.

Table 7-17. Average concentrations (mg/kg) of selected metals in sediment from the Little Scioto River, Ohio, by river mile in 1987, 1991,1992 and 1998.*

Metal	River Mile								
Arsenic	[11.1]	/9.4\ (9.5) [9.2]	(7.9)	/7.2\ [7.1]	{6.5} /6.6\ (6.5) [6.6]	/5.8\ (5.8) [6.2]	(4.4)	{2.7} /2.7\ (2.67) [2.65]	(0.36)
	[3.6]J	/<10\ (<10) [8.3]J	(12.4)	/<10\ [6.0]J	{11.2} /<10\ (<10) [10.8]J	/<10\ (13.8) [9.8]J	(11.3)	{9.49} (<10) [9.0]	(<10)
Cadmium	[0.1]ND	/<1.0\ (<1.0) [0.1]ND	(<1.0)	/<1.0\ [0.2]	{1.8} /3.4\ (<1.0) [0.1]ND	/1.0\ (<1.0) [2.0]	(10.5)	{4.39} (1.0) [1.4]	(1.6)
Chromium	[8.1]J	/5.8\ (7.3) [14.3]J	(13.6)	/13.2\ [8.9]	{47.6} /415\ (208) [32.3]J	/39.2\ (60.9) [50.4]	(302)	{134} (71.2) [77.1]	(48.6)
Copper	[15.7]	(7.4) [24.1]	(17.2)	[22.9]	{68} (79) [39.2]	(56.0) [133]	(76.8)	{83} (42.4) [79.3]	(24.5)
Lead	[23.8]	/<10\ (12.1) [20.4]	(19.1)	/25.5\ [24]J	{170} /175.5\ (172) [46.4]	/59.5\ (84.6) [220]J	(93.4)	{160} (108) [180]J	(38)
Mercury	[0.1]ND	/<0.1\ (<0.1) [0.2]J	(<0.1)	/<0.1\ [0.1]J	/0.3\ (0.33) [0.3]J	/0.2\ (0.2) [0.6]J	(0.8)	(0.12) [0.4]J	(<0.1)

Table 7-17 (continued). Average concentrations (mg/kg) of selected metals in sediment from the Little Scioto River, Ohio, by river mile in 1987, 1991,1992 and 1998.*

Metal	River Mile								
	[11.1]	/9.4\ (9.5) [9.2]	(7.9)	/7.2\ [7.1]	{6.5} /6.6\ (6.5) [6.6]	/5.8\ (5.8) [6.2]	(4.4)	{2.7} /2.7\ (2.67) [2.65]	(0.36)
Zinc	[48.2]	(30.6) [81.4]	(79.0)	[66.6]	{187} (173) [89.2]	(141) [280]J	(226)	{760} (408) [316]J	(96.8)

{ } = 1987 data from OEPA 1988, sample depth unknown
/ \ = 1991 data from OEPA 1992, sample depth unknown
( ) = 1992-93 data from OEPA 1994, sample from 1-6" except RM 7.9 sample from 8-12"
[ ] = 1998 data from OEPA unpublished, sample depth unknown
J is an estimated value that is above zero but below the practical quantitation limit.

Table 7-18. Average concentrations of selected water chemistry parameters (mg/L) in the Little Scioto River, Ohio, by river mile in 1987, 1992 and 1998.*

Compound	**River Mile**								
	[11.1]	(9.2) [9.2]	{7.9} (7.9)	[7.1]	{6.5} (6.5)	{5.8} (5.8) [6.2]	{4.4} (4.4)	{2.7} (2.7) [2.7]	{0.4} (0.4)
Ammonia	[0.1,0.3]	(<0.05) [<0.05,<0.0 5]	(<0.05)	[0.11, <0.05]	(0.12)	(1.16) [0.35, 0.69]	(1.44)	(2.10) [0.67, 1.1]	(0.58)
Dissolved oxygen**		(12.2, 8.8)	{4.6, 2.8} (7.9, 5.7)		{7.27, 1.9}	{8.3, 4.2} (8.23, 4.21)	{8.8, 3.2} (5.2, 4.3)	{6.67, 2.0} (4.1, 3.0)	{6.74, 2.5} (5.6, 4.4)
BOD	[<2.0, 6.6]	(1.0) [<2.0, <2.0]	(1.0)	[<2.0, 2.1]	(2.3)	(4.7) [4.6,13]	(4.2)	(3.5) [3.3, 4.1]	(2.2)
Nitrate-nitrite, NO_x	[0.7,3.3]	(1.2) [0.4, 0.2]	(1.4)	[0.73, <0.1]	(0.8)	(8.1) [0.33, 2.37]	(6.6)	(4.5) [3.5, 0.9]	(4.47)
Phosphorus, total P	[0.5,0.6]	(0.06) [1.8, 0.1]	(0.07)	[0.36, 0.13]	(0.09)	{1.65} (2.17) [1.9, 1.21]	(1.96)	{2.71} (1.80) [1.18, 1.31]	(1.34)
Hardness, $CaCO_3$	[222,250]	(329) [275, 269]	(327)	[281, 407]	(389)	(278) [224, 261]	(280)	(306) [228, 210]	(320)

* { } = 1987 (OEPA 1988b; ( ) = 1992-1993 (OEPA 1994) [ ] = 1998 (OEPA August and October, unpublished data).
** Dissolved Oxygen {maximum, minimum}, data from 1987 (OEPA, 1988b).
(maximum, minimum from box plots), data from 1992 (OEPA, 1994.

Table 7-19. PAH concentrations at nearest upstream location and locations of impairments (mg/kg). (*Hyalella azteca* sediment effects concentrations, PEL and TEL, normalized to sediment WET weight.)

Chemical		PAH sediment concentration			
PEL	TEL	Nearest Upstream Location	Impairment A	Impairment B	Impairment C
Benzo(a)pyrene (BAP) 0.32	0.03	(0) [0.053] #	(0) [0.043] #	/141.1\ * (14.8) * [210] *	/125\ * (15.8) * [14] *
Naphthalene (NAPH) 0.14	0.02	(0) [0]	(0) [0]	/22.9* (0) [260]*	/70\ * (4.6) * [18] *
Fluorene 0.15	0.01	(0) [0]	(0) [0.059] #	(0) [830] *	/200\ * (7) * [20] *
Phenanthrene 0.41	0.02	(0) [0.053] #	(0) [0.11] #	(0) [230] *	/470\ * (24.1) * [38] *
Anthracene 0.17	0.03	(0) [0]	(0) [0.037] #	(0) [100] *	/360\ * (27.1) * [41] *
Fluoranthene 0.32	0.04	(0) [0.097] #	(0) [0.2] #	(8.2) * [100] *	/550\ * (37.6) * [44] *
Pyrene 0.49	0.02	(0) [0.076] #	(0) [0.16] #	(0) [810] *	/405\ * (23.8) * [32] *
Benzo[a]anthracene 0.28	0.03	(0) [0.043] #	(0) [0.059] #	(8.2) * [310] *	/185\ * (16.5) * [42] *
Chrysene 0.41	0.02	(0) [0.065] #	(0) [0.081] #	(16.5) * [390] *	/305\ * (20.8) * [13] *
Benzo(g,h,i)perylene 0.25	0.01	(0) [0.044] #	(0) [0.03] #	(49.5) * [150] *	/65\ * (11.2) * [19] *

(*) exceeds PEL and TEL; (#) exceeds TEL. / \ = 1987-1991, ( ) = 1992, [ ] = 1998.
Zero = below detection; No Entry = No data for that year

Table 7-20. Metals concentrations at nearest upstream location and locations of impairments (mg/kg). (*Hyalella azteca* sediment effects concentrations, PEL and TEL, normalized to sediment wet weight.)

Chemical		Nearest Upstream Location	Impairment A	Impairment B	Impairment C
PEL	TEL				
As 48.4	10.8	/5\ (5) [8.3]	/8\ (12.4) # [6]	/11.2\ # (0) [10.8] #	/8\ (13.8) # [9.8]
Cd 3.2	0.58	/0.5\ (0.5) [0]	/0.5\ (0.5) [0.2]	/1.8\ # (0.5) [0.1]	/1\ (0.5) [2] #
Cr 119.4	32.3	/5.8\ (7.3) [14.3]	/13.2\ (13.6) [8.9]	/47.6\ # (208) * [32.3] #	/39.2\ # (60.9) # [50.4] #
Cu 101.2	28	(7.4) [24.1]	(17.2) [22.9]	/68\ # (79) # [39.2] #	(56) # [133] *
Pb 81.7	37.2	(12.1) [20.4]	(19.1) [24]	/170\ * (172) * [46.4] #	/59.5\ # (84.6) * [220] *
Zn 544	98.1	(30.6) [81.4]	(79) [66.6]	/187\ # (173) # [89.2]	(141) # [280] #

(*) exceeds PEL and TEL; (#) exceeds TEL. *ND= not detected, NA = not available, / \ = 1987-1991, ( ) = 1992, [ ] = 1998. Zero = below detection; No Entry = No data for that year

APPENDIX A
OVERVIEW OF WATER MANAGEMENT PROGRAMS SUPPORTED BY THE SI

Appendix A

Overview of Water Management Programs Supported by the SI

The following sections describe several major water management programs and how the SI process can support them.

A.1 Water Quality Assessment Reports Under CWA Section 305(b)

In 1987, EPA's Office of Water recommended that regulatory authorities increase the use of biological monitoring to better characterize aquatic systems. State and Tribal agencies were directed to protect the fishable and swimmable goals of the Clean Water Act. Under Section 305(b), States, Territories, the District of Columbia, interstate water commissions, and participating American Indian Tribes are required to assess and report on the quality of their waters (USEPA 1997). The results of 305(b) assessments are not raw data, but rather are statements about the degree to which each waterbody supports the uses designated in state or tribal water quality standards. Each State and Tribe aggregates these assessments and extensive programmatic information in a 305(b) report, which is a detailed document usually including information from multiple agencies. EPA then uses individual 305(b) reports to prepare a biennial National Water Quality Inventory Report to Congress. This report is the primary vehicle for informing Congress and the public about water quality conditions in the United States.

Most of the information contained in 305(b) assessments is based on data collected and evaluated by states, tribes, and other jurisdictions over the two-year period immediately preceding issuance of the report. The Report to Congress contains national summary information about water quality conditions in rivers, lakes, estuaries, wetlands, coastal waters, the Great Lakes, and groundwater. The report also contains information about public health and aquatic ecosystem concerns, water quality monitoring, and state and federal water pollution management programs.

States and Tribes base their 305(b) water quality determinations on whether waterbodies are clean enough to support basic uses, such as aquatic life, swimming, fishing, and drinking supply. These uses, along with appropriate national criteria and anti-degradation statements, are part of the water quality standards set by each state or tribe to protect its waters. These standards must be approved by EPA.

Water quality for each individual use is rated as either:

- Good/Fully Supporting
- Good/Threatened
- Fair/Partially Supporting
- Poor/Not Supporting
- Poor/Not Attainable

For waterbodies with more than one use, information is consolidated into a summary use support designation of general water quality conditions. These uses are characterized as either:

- Good/Fully Supporting All Uses
- Good/Threatened for One or More Uses
- Impaired for One or More Uses

Once a state or tribe has determined, under section 305(b), that a waterbody is impaired for one or more uses, the state or tribe is required to identify the source and cause of impairment. Some causes are much easier to identify than others. For example, a case where impairment is caused by a specific chemical from a point source discharge might be straightforward and easily analyzed. Monitoring programs, however, must deal with impacts caused not only by chemical toxicity, but also conventional pollutants (e.g., temperature, pH and dissolved oxygen) and anthropogenic pollutants from non-point sources. Monitoring agencies need the ability to evaluate the relative impact that a particular pollutant or other stressor has on the biological integrity of a receiving water.

A.2 303(d) Lists and TMDLs

Section 303 of the 1972 Clean Water Act requires States, Territories and authorized Tribes to establish water quality standards and Total Maximum Daily Loads TMDLs) for EPA review and approval. Water quality standards identify the uses for each waterbody (e.g., drinking water supply, contact recreation, aquatic life support) and the water quality criteria to support that use. Water quality criteria can be either numeric (e.g., no more than 10 μg/L of copper) or narrative (e.g., nutrients are not to exceed levels which cause an imbalance of aquatic flora and fauna). Water quality standards also include antidegradation policies to prevent deterioration of existing high quality waters.

Under Section 303(d), States, Territories and authorized Tribes must identify impaired waters and establish TMDLs for these waters. Impaired waters are those that do not meet applicable water quality standards, even after point sources of pollution have installed the minimum required levels of pollution control technology. States, Territories and authorized Tribes are required to submit their list of impaired every two years.

States, Territories and authorized Tribes are required to establish priority rankings for impaired waters on the 303(d) lists and develop TMDLs for these waters. A TMDL specifies the maximum amount of a pollutant that a waterbody can receive and still meet water quality standards, and allocates pollutant loadings among point and nonpoint pollutant sources. EPA must approve or disapprove lists and TMDLs established by States, Territories and authorized Tribes. If a State, Territory or authorized Tribe submission is inadequate, EPA must identify the impaired waters and establish the TMDL.

TMDLs are a critical component of the water quality program. They provide the analytic underpinning for watershed decisions and promote integrated program planning, implementation, and funding. For example, controlling sediment and/or nutrient loadings can protect aquatic habitat, wetlands, endangered species, and drinking water sources. As requirements are strengthened and public communication emphasized, sound procedures for identifying stressors and management solutions will become more important.

Development of a TMDL varies based on numerous factors including environmental setting, waterbody type, source type/behavior, and pollutant type/behavior. However, TMDL development generally includes the following activities:

1. Problem Identification: characterization of the impairment and identification of the pollutant causing the impairment;

2. Identification of Water Quality Targets: establishment of the TMDL endpoint or target value, which is typically the applicable numeric water quality criterion or a numeric interpretation of the narrative water quality standard;

3. Source Assessment: estimation of the point, nonpoint and background sources of pollutants of concern, including magnitude and location of sources;

4. Allocations: identification of appropriate wasteload allocations for point sources and load allocations for nonpoint sources;

5. Link Between Numeric Target(s) and Pollutant(s) of Concern: Analysis of the relationship between numeric target(s) and identified pollutant sources. For each pollutant, describes the analytical basis for conclusion that sum of wasteload allocations, load allocations, and margin of safety does not exceed the loading capacity of the receiving water(s).

6. Calculation of the explicit or implicit margin of safety for each pollutant and description of accounting for seasonal variations and critical conditions in the TMDL.

A.2.1 Causes for Impairment: Pollutants and Pollution

Waterbodies are impaired by a variety of stressors. Recent data indicate that the top causes for impairment include sedimentation/siltation/turbidity and suspended solids (16%), nutrients (13%), pathogens (13%), and dissolved oxygen (10%). These stressors are often associated with sources or activities that fall under the Clean Water Act definition of *pollutant*, or *pollution*. Pollution is defined in Section 502(19) as the “man-made or man-induced alteration of the chemical, physical, biological, and radiological integrity of water.”

Section 303(d) requires the identification and listing of all impaired waterbodies regardless of the origin or source of the pollution or pollutant. Current regulations require that TMDLs be calculated only for *pollutants*. Pollutants are defined in Section 502(6) as “dredged spoil, solid waste, incinerator residue, sewage, garbage, heat, and industrial, municipal, and agricultural waste discharged into water.”

Both *pollution* and *pollutants* are “stressors” that can be identified and evaluated using the SI process. Under current regulations, those calculating TMDLs will benefit directly from guidance on identifying stressors considered *pollutants* under the Clean Water Act. The SI guidance can also assist in establishing the causal linkage between a pollutant and the biological impairment, and thus provide a basis for the development of a TMDL. For example, if a pollutant causes ecosystem changes that alter the fish community, the altered biological community is an impairment that can be traced to a pollutant for which a TMDL can be calculated.

A.2.2 EPA Actions to Implement the TMDL Program

In an effort to speed the Nation's progress toward achieving water quality standards and improving the TMDL program, EPA began, in 1996, a comprehensive evaluation of EPA's and the states' implementation of their Clean Water Act section 303(d) responsibilities. EPA convened a committee under the Federal Advisory Committee Act, composed of 20 individuals with diverse backgrounds, including agriculture, forestry, environmental advocacy, industry, and state, local, and tribal governments. The committee issued its recommendations in 1998. These recommendations were used to guide the development of proposed changes to the TMDL regulations, which EPA issued in draft in August, 1999. After a long comment period, hundreds of meetings and conference calls, much debate, and the Agency's review and serious consideration of over 34,000 comments, the final rule was published on July 13, 2000. However, Congress added a "rider" to one of their appropriations bills that prohibits EPA from spending FY2000 and FY2001 money to implement this new rule. The current rule remains in effect until 30 days after Congress permits EPA to implement the new rule. TMDLs continue to be developed and completed under the current rule, as required by the 1972 law and many court orders. The regulations that currently apply are those that were issued in 1985 and amended in 1992 (40 CFR Part 130, section 130.7). These regulations mandate that states, territories, and authorized tribes list impaired and threatened waters and develop TMDLs.

A.2.3 Stressor Identification and the TMDL Program

EPA developed the SI process to assist water resource managers in identifying and delineating stressors causing biological impairments to waterbodies. While not all water quality impairments listed under 303(d) are linked directly to biological components of waterbodies, a sample of submittals from 19 states indicate that approximately one-half of waterbodies listed as impaired under 303(d) are not meeting biological designated uses (e.g., aquatic life, cold water fishery). The SI process will have direct utility to States, Tribes, and EPA by providing sound approaches to evaluating the causes of biological impairments under the TMDL Program.

As used in the SI process, the term *stressor* is synonymous with the terms *pollutant* and *pollution* which, under Section 303(d), are considered causes of impairment. The identification of *pollutant* stressors resulting in biological impairment to waterbodies, and the diagnostic evaluation of the sources of these stressors, is an essential first step in calculating Total Maximum Daily Loads under Section 303(d) of the Clean Water Act. For *pollution* stressors (e.g., habitat degradation, water control structures), for which TMDLs are not calculated, SI results can be used to identify the sources of the pollution for use in alternative watershed management activities.

A.3 State/Local Watershed Management

Since 1991, EPA has promoted a watershed protection approach to help address the nation's remaining water resource challenges (USEPA 1991a). The watershed approach is an integrated, holistic strategy for protecting and managing surface water and groundwater resources by watershed, a naturally defined hydrologic unit. For any given watershed, the approach considers not only the water resource; such as a stream, river, lake, estuary, or aquifer; but all of the land from which water drains into that resource. The watershed approach uses all aspects of water resource quality—physical (e.g., temperature, flow, mixing, habitat); chemical (e.g., conventional and toxic pollutants, such as nutrients and pesticides); and biological (e.g., health and integrity of biotic communities, biodiversity). EPA's Office of Water has worked to orient and coordinate point source, non-point source, surface water, wetlands, coastal, groundwater, and drinking water programs within a watershed context.

The watershed approach is not a program but a way to organize programs, so that the use of SI will vary with the program conducting the investigation. The watershed approach, however, can facilitate an SI investigation since information is already integrated from various sources, such as point source discharges and non-point source runoff. This integrated information can help investigators make sense of disturbances through knowledge of potential sources of stressors that might feed into that location or might affect the food source or some other essential ecosystem component by affecting the natural continuum (Vannote et al. 1980).

The challenge for identifying stressors for watershed-based programs is proper scaling. Even though the SI may be initiated by a program using the watershed approach, the impairment may not be watershed wide. Impairment to the biological system may be difficult to determine on a watershed scale. Similarities among biota tend to follow ecoregions, rather than watersheds. Several ecoregions may exist within a watershed, especially where elevation differences are great. The biota within any given ecoregion may respond differently to a given stressor than the biota within a neighboring ecoregion. Accurate scaling of the problem is important any time a biological impairment is found, but especially with the watershed approach, to ensure that the information is used to full advantage in identifying and characterizing stressors.

A.4 Non-point Source 319 Management

The 1987 Water Quality Act Amendments to the Clean Water Act added section 319, which established a national program to assess and control non-point source (NPS) pollution. Under this program, states and tribes are asked to assess their NPS pollution problems and submit their assessments to EPA. The assessments included a list of navigable waters within the State or Tribal Territories, which without additional action to control NPS pollution, cannot reasonably be expected to attain or maintain applicable water quality standards or the goals and requirements of the Clean Water Act. Section 319 also requires identification of categories and subcategories of NPS pollution that contribute to impairment of waters, descriptions of procedures for identifying and implementing best management practices, control measures for reducing NPS pollution, and descriptions of State, Tribal, and local programs used to abate NPS pollution.

NPS programs need to identify and control NPS pollutants. Since NPS pollutants can be difficult to trace, identifying the source of these pollutants is probably the greatest challenge for NPS programs. The SI process can help investigators obtain greater confidence that stressors have been accurately identified. Attributing responsibility to a particular source can be very straightforward and obvious or very difficult. Mechanisms used to attribute responsibility need to be assessed for each situation, and common sense should be used. For example, runoff may be obviously coming from one farm. In another situation, runoff may encounter multiple potential sources of pollution, including a poultry farm, a cattle feedlot, and an abandoned mine. In the latter situation, if nutrient loading is the identified stressor, attributing responsibility between the poultry farm and cattle feedlot may be difficult, but ruling out the abandoned mine would be simple.

A.5 Permitting Programs

A.5.1 NPDES Permits

All discrete sources of wastewater are required to obtain a National Pollutant Discharge Elimination System (NPDES) permit (or State equivalent) that regulates the facility's discharge of pollutants. This approach to controlling and eliminating water pollution is focused on pollutants determined to be harmful to receiving waters and sources of such pollutants. Authority for issuing NPDES permits is established under Section 402 of the CWA. A summary of the Water Quality-based "Standards to Permits" Process for Toxics Control (adapted from the Technical Support Document for WQ-based Toxics Control, TSD, USEPA 1991a) lists nine steps:

1. Define water quality objectives, criteria, and standards;

2. Establish priority waterbodies;

3. Characterize effluent - chemical-specific or Whole Effluent Toxicity (WET);
   a) evaluate for excursions above standards,
   b) determine reasonable potential, and
   c) generate effluent data;

4. Evaluate exposure (critical flow, fate modeling, and mixing) and calculate wasteload allocation;

5. Define required discharge characteristics by the waste load allocation;

6. Derive permit requirements;

7. Evaluate toxicity reduction and/or investigate indicator parameters (as needed, for permits containing WET monitoring or limits);

8. Issue final permit with monitoring requirements – average monthly and maximum daily average weekly for publicly operated treatment works) limits; and

9. Track compliance.

Sometimes the monitoring requirements include biological assessment of the receiving water. The permit can contain a reopener clause to allow the limits and monitoring requirements to be adjusted if biological impairment is found in the receiving water.

The SI guidance is somewhat analogous in function to the Toxicity Reduction Evaluation (TRE) and Toxicity Identification Evaluation (TIE) guidance used in Step 7 above (USEPA 1988a,b,c, 1991b, 1993a,b). In the permitting process, toxicity is controlled through limits for specific chemicals and limits for whole effluent toxicity. When permit monitoring shows that an effluent has toxicity above the amount allowed by the permit, the discharger is often required to conduct a TRE to determine if a simple solution exists for reducing the toxicity, e.g., housekeeping procedures for cleaning fluids, or pH buffering of the effluent. If the solution is not apparent from the TRE, additional TIE procedures may be required. TIE procedures guide investigators through additional data collection to determine the toxic component(s) of the waste stream. These procedures include both aquatic toxicity methods and chemistry methods.

When WET or chemical testing show that the effluent is toxic, this does not mean that an impairment will necessarily be found in the aquatic biota within the zone of influence of the discharge. Effluent limits include safety factors in their calculations. The waste load allocation (Step 4, above) is calculated based on worst-case estimations. For example, effluent limits for toxicity or for a toxic chemical are based on low-flow conditions in streams and rivers (often the lowest seven-day flow in a ten-year period). Effluent limits may be exceeded, a TRE/TIE conducted, and the problem solved without incurring measurable impairment in the receiving water biota. The current trend is to lessen this safety buffer by customizing water quality-based permit limits to local conditions through such mechanisms as dynamic modeling of waste load allocation (USEPA 1991a) and recalculation of water quality standards or use of the water-effects ratio (USEPA 1994).

Conversely, ambient biological assessments may show impairment in the aquatic biota below a permitted discharge without a measured permit limit exceedence. The role of the effluent in causing the impairment is not readily apparent in this case. The effluent stream could have been toxic during periods when toxic parameters were not being measured; effluent toxicity tests could have been insufficiently sensitive through inappropriate selection of test organisms or operator error; or impairment could have been caused by stressors other than effluent discharge. Accurate attribution of responsibility can be very critical in NPDES permitting cases, both for fairness and success in stressor control. A SI should be conducted to distinguish effects caused by the effluent discharge and effects from other stressors.

A.5.2 Cooling Tower Intake 316(b) Permitting

Under section 316(b) of the CWA, any NPDES permitted discharger which intakes cooling water must not cause an adverse environmental impact to the waterbody. To determine if a cooling water intake structure is causing adverse environmental impacts to the waterbody, the overall health of the waterbody should be known. Where biological impairments are found, stressor identification procedures should help investigators identify the different stressors causing the waterbody to be impaired, including the intake structure. A high degree of certainty is needed.

A.5.3 Dredge and Fill Permitting

Under Section 401 of the CWA, different types of federal permitting activities (such as wetlands dredge and fill permitting) require a certification that there will be no adverse impact on water quality as a result of the activity. This certification process is the 401 Water Quality Certification. Under Section 404 of the CWA, the discharge of dredge and fill materials into a wetland is illegal unless authorized by a 404 Permit. The 404 Permit must receive a 401 Water Quality Certification.

Stressor identification procedures will help investigators identify the different types of stress an activity may place on water quality that can then be addressed through conditions in the 401 Certification. Stressor identification procedures may help to identify unanticipated stress from a dredge and fill activity on water quality or the biological community after the activity is underway. Stressor identification procedures may also help in pre-permitting evaluations of the potential impacts of 404 permitting by assessing different potential stressors on the wetland in advance.

A.6 Compliance and Enforcement

Since 1972, Section 309 of the Clean Water Act has provided statutory authority for a range of enforcement responses for entities or individuals who fail to comply with the Act. At the extreme end of this range, actions can result in criminal penalties. EPA has national and regional programs in place to investigate and prosecute cases. States and Tribes may have their own compliance and enforcement investigation programs.

A.6.1 Investigations

When a violation occurs, an investigator must first ascertain what must be done to achieve compliance with the Clean Water Act. Under a Section 309 order, the violator must come in full compliance with the Clean Water Act; which, under Article 101, directs the restoration and maintenance of the biological integrity of the nation's waters. When non-compliance is due to biological impairment or non-attainment of biological integrity, the investigator must determine the cause of the impairment before implementing a program to restore biological integrity and achieve compliance. This is a direct use of the SI process.

The degree of environmental harm is a very important factor that investigators and judges evaluate when assessing criminal penalties. The SI process should be helpful in determining whether the causes of impairment are consistent with the causes that would likely have resulted from the source under investigation. The SI process can also help to determine the likelihood that one stressor versus another caused the impairment. In cases where separation of stressor mechanisms is fairly clear cut, the SI process can help investigators determine the significance of the available evidence in determining whether the alleged stressor caused the noted environmental harm. However, the SI process is limited to evaluating causes. If more than one stressor or source are involved, allocating the relative contribution of each stressor or source to the environmental harm may require additional tools, such as allocation methodologies, that are beyond the scope of this document.

A.6.2 Enforcement Proceedings

In an enforcement action, the enforcement official seeks for a court to order the defendant to cease the harmful action, or give injunctive relief. Identifying the causes of impairment is a crucial step in identifying the actions that would constitute injunctive relief. The SI process should benefit enforcement officials and expert witnesses by helping them identify responsible stressors and organize cogent evidence supporting the identified causal scenario. The SI process adds uniformity to the organization and analysis of data.

A special program that is often used to grant injunctive relief is the Supplemental Environmental Project (SEP). Under this program, a judge may allow a defendant to improve the environment in lieu of paying a portion of a federal fine to the National Treasury. The environmental benefit gained through an SEP may not directly alter the harm that the defendant caused originally, but is seen as alternate compensation. For example, rather than paying a fine of $1 million, a defendant might pay a $600,000 fine and build a bike path with a 30-foot riparian buffer zone (for runoff reduction) along the impacted creek, or even a neighboring stream.

When the SI process identifies multiple stressors as the cause of impairment, the information can still be valuable to the SEP program because the alternate stressors may help direct compensatory action. If, for example, the SI process identifies a stressor scenario with two stressors working in conjunction and the defendant is responsible for only one of the two stressors, a judge might approve a plan for the defendant to use resources to conduct an SEP project that reduces the second stressor, in lieu of a portion of the fine.

Targeting resources is very important to investigation and enforcement efforts. EPA often uses 303d lists of impaired waterbodies to target these efforts. The SI process can supplement the information in the 303d lists so that stressors may be targeted within targeted waterbodies. Targeting may also be important in assessing future legislative needs when mechanisms for stressor control are inadequate in national rules and policies, and in current state and tribal statutes. Targeting stressors for increased control may identify changes to instigate.

A.7 Risk Assessment

Risk assessment is a scientific process that includes stressor identification, receptor characterization and endpoint selection, exposure assessment, stress-response assessment, and risk characterization (USEPA 1998a, Suter 1993). Risk management is a decision-making process that combines human-health and ecological assessment results with political, legal, economic, and ethical values to develop and enforce environmental standards, criteria, and regulations. Risk assessment can be performed on a site-specific basis, or can be geographically-based (e.g., watershed scale). It can be used to assess human health or ecological risks.

Results of bioassessment studies can be used in watershed ecological risk assessments to develop broad-scale empirical models of biological responses to stressors. Such models can be combined with exposure information to predict risk from specific stressors and anticipate the success of management actions. Accurate stressor identification is an integral part of this process and can help ensure that management actions are properly targeted and efficient in producing the desired results.

A.8 Wetlands Assessments

Although few states have fully incorporated wetlands into water quality standards or biological assessment programs, a growing number have started to develop biological assessment methods for wetlands. During the past five years, several state and federal agencies have independently started to develop bioassessment methods for wetlands. Minnesota, Montana, North Dakota, and Ohio have been pioneers among the states. The Biological Resource Division of the U.S. Geological Service, Wetlands Science Institute of the Natural Resources Conservation Service, and EPA have been the leading federal agencies.

The SI process and tools specific to wetlands investigations are very much needed by wetlands managers. In recent 305(b) Reports, states identified sedimentation, nutrient enrichment, fill and drainage, pesticides, and flow alterations as the major causes of wetlands degradation. Biological assessment methods will allow resource managers to evaluate the condition of wetlands and may provide some indication of the types of stressors involved. Once bioassessment methods are completed and incorporated into monitoring programs, wetlands may be listed as impaired due to biological impairment. SI methods will be needed to identify stressors causing biological impairment so that resource managers can better remedy the problems. More information about wetland bioassessments is available at the EPA Wetlands Division web page (www.epa.gov/owow/wetlands).

A.9 Preservation and Restoration Programs

Preservation and restoration programs like the National Estuary Program and the Superfund Program can also benefit from the SI process.

A.9.1 National Estuary Program

The National Estuary Program (NEP) was established in 1987 by amendments to the Clean Water Act to identify, restore, and protect nationally significant estuaries of the United States. Unlike traditional regulatory approaches to environmental protection, the NEP targets a broad range of issues and engages local communities in the process. The program focuses not only on improving water quality in an estuary, but also on maintaining the integrity of the whole system, its chemical, physical, and biological properties, and its economic, recreational, and aesthetic values.

The NEP is designed to encourage local communities to take responsibility for managing their own estuaries. Each NEP is made up of representatives from federal, state and local government agencies responsible for managing the estuary's resources, as well as members of the community -- citizens, business leaders, educators, and researchers. These stakeholders work together to identify problems in the estuary, develop specific actions to address those problems, and create and implement a formal management plan to restore and protect the estuary. Twenty-eight estuary programs are currently working to safeguard the health of some of our nation’s most important coastal waters.

The SI process should be useful to the NEP, and other preservation programs, by helping stakeholders identify sources and causes of impairments. This information would feed into the development of a management plan.

A.9.2 Superfund

The Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), commonly known as Superfund, was enacted in 1980 (and amended in 1986) for hazardous waste cleanup. This law created a tax on the chemical and petroleum industries and provided federal authority to respond to releases or threatened releases of hazardous substances that may endanger public health or the environment. The money collected from the taxation went to a trust fund for cleaning up abandoned or uncontrolled hazardous waste sites. CERCLA also established prohibitions and requirements for closed and abandoned hazardous waste sites; defined liability of persons responsible for releases of hazardous waste at these sites; and established funding for cleanup when no responsible party could be identified.

Since the basis for actions is whether the hazardous substance may endanger public health or the environment, identifying the stressor(s) causing environmental harm is important. For cleanup sites where other stressors (e.g., habitat alteration) are also likely causes of impairment, any cleanup and ecosystem recovery plans would need to take into account the effects of these stressors. Allocating the amount of responsibility that may be attributed to each stressor is beyond the scope of the SI process, but knowledge of any additional stressors that may be causing effects can be valuable in determining expected outcomes of recovery activities.

APPENDIX B
WORKSHEET MODEL

Appendix B

Worksheet Model

The following pages contain a worksheet model that may be used with the SI process. This is only an example and may not fit every case without alterations.

B.1 Instructions for Using the Worksheet Model

This worksheet follows the SI process outlined in this document. The worksheet was designed to be flexible. At certain points, the user will be asked to stop () and consider the evidence gathered thus far, in order to determine whether the process is complete or requires further analysis. For detailed guidance, the user will need to refer to the sections of the document that are cited at each step.

1. To begin, write the name of the investigator and date for reference.

2. Fill in the appropriate information in *Unit I: List Candidate Causes*. To determine the types of information to include throughout the worksheet, please refer to the cited sections of the document.

3. Summarize and document the data and analyses in *Unit II, Part A*. Then, you may use either of the following options:

   - Option 1: Analyze the strongest evidence. If you feel that you have enough **case specific** data to eliminate some causes, analyze this data using *Unit II, Part B* and proceed to *Unit III, Step 1: Eliminate Alternatives*. Note: You may also look at other types of evidence that can be used for elimination in *Unit II, Parts C and D.* To do this, fill in only the blanks in Parts C and D that are designated by the letter **E** (for elimination) under the heading *Associated Causal Characterization Method in Unit III*. Review this additional evidence to see if it allows you to eliminate any alternatives.

   - If you still have more than one likely causal scenario that could not be readily eliminated, or if you want to thoroughly review all evidence, proceed to *Unit II, Parts C and D.* Complete relevant sections of *Parts C and D* for each candidate cause that you listed in *Unit I.* Then proceed to *Unit III* and characterize the cause using diagnosis or strength of evidence, as appropriate (described under #4 below).

   - Option 2: List all available evidence in *Unit II* before going on to *Unit III: Characterize Causes*. Using either option, you may still choose to do additional iterations if the available evidence is insufficient.

   - Go to *Unit III, Characterize Causes*. For those candidate causes listed in *Unit I* that were not eliminated while analyzing the evidence listed in *Unit II* (i.e., those causes not designated as ***E*** in *Parts C and D* under the heading *Associated Causal Characterization Method in Unit III*), complete *Step 1: Eliminate Alternatives* and try to further eliminate causes. **Analyze this evidence carefully**; if the evidence is not strong enough to eliminate a candidate, it still may be useful for the strength of evidence analysis. Using the worksheet in *Unit III, Step 1*, determine:

- If the primary cause is so dominant that it masks the effects of others, then re-evaluate whether the other stressors should be retained. A cause should not be eliminated if it is potentially masked. Instead, strength of analysis should be used.

- If only one candidate cause remains, go to *Unit IV: Sufficiency of Evidence*. Note: You still may want to look at the diagnostic and strength of evidence information to strengthen your case. If so, go to *Unit III*, *Step 2*.

- If more than one candidate cause remains, go to *Unit III*, *Step 2* to look for diagnostic evidence.

- If no candidate causes remain, go to *Unit V*. You will need to do another iteration with more information.

- Next, try diagnosis. Look for evidence designated as ***D*** under the column labeled *Associated Causal Characterization Method in Unit III* in *Unit II*, *Part C* tables. Using the worksheet in *Unit III*, *Step 2*, determine:

- If only one candidate cause remains, go to *Unit IV: Sufficiency of Evidence*. Note: You may still want to do a strength of evidence analysis to strengthen your case. If so, go to *Unit III*, *Step 3*.

- If more than one candidate cause remains, go to *Unit III*, *Step 3* (*Strength of Evidence Analysis*).

- If no candidate causes remain go to *Unit V* and do another iteration with more information.

- Many investigators will want to complete the strength of evidence analysis even if elimination or diagnosis have identified the stressor. **This part of the SI process helps determine how strong a case an investigator can make for a particular stressor.** Look for evidence designated as ***S*** under the column labeled *Associated Causal Characterization Method in Unit III* in *Unit II Part C*, and also consider the evidence gathered in *Part D*. Analyze this evidence carefully using the worksheet in *Unit III*, *Steps 3, 4, and 5*.

- *Unit III Steps 3*, *4*, and *5* allow the investigator to compare evidence, side-by-side, for candidate causes. The step used depends on the type of evidence. Scores are assigned to each candidate cause to reflect that cause's relevance to each causal consideration. (For more detailed information on comparing stressors, refer to the sections cited in the worksheets). Compare scores among the candidate causes, and then go to *Unit IV*, *Sufficiency of Evidence*.

- List the most likely cause in *Unit IV*, and determine if the evidence is sufficient for the intended use.

- If yes, your SI is complete, report results.

- If no, go to *Unit V, Reconsider Impairment*.

- Reconsider whether the impairment was real and describe the results.

- If no, your SI is complete, report results.

- If yes, go to *Unit VI, Collect More Data*.

- Determine whether all reasonable causes were analyzed.

- If no, complete *Unit VI, Follow-on 1* to determine whether additional scenarios should be analyzed (back to *Unit I*), or whether the process should be ended and the results reported as inconclusive.

- If yes, go to *Unit VI, Follow-on 2* to determine whether additional data should be collected and another iteration begun (back to *Unit I*), or whether the process should be ended and the results reported as inconclusive.

Stressor Identification Worksheet

Investigator ______________________________ Date Completed_________________

UNIT I. LIST CANDIDATE CAUSES	
	Results / Notes
Describe the impairment. *(see Chapter 2.2)*	
Make a map. (Unit I part A) *(see Chapter 2.2)*	
Define the Scope of the Investigation. *(see Chapter 2.3)*	
List the candidate causes (*see Chapter 2.4)*	
Develop a conceptual model for the case. (Unit I, part B) *(see Chapter 2.5)*	
Candidate Causes	
# 1.	
# 2.	
# 3.	

Go to Unit II, Analyze Evidence.

UNIT I. LIST CANDIDATE CAUSES

Part A. Make a map to document geographic features relevant to the analysis.

- Draw a map or insert map of study area.
- Include natural and man-made features such as dams, sources, tributaries, landfills, dredge areas, jetties, sand bars, waterfalls, wetlands, salt water intrusion, etc. See Chapter 2.2.
- Show location of impairment.

UNIT I. LIST CANDIDATE CAUSES

Part B. Make a conceptual model of the case.

- Draw a conceptual model of the case. See Chapter 2.5.
- Include hypothesized sources, stressors and important environmental processes that lead to the impairment.
- Label candidate causes.

UNIT II
Part A. Summarize and document associations between the candidate cause and the effect from the case. • Insert tables, graphs and/or figures of relevant data. See Chapter 3.1. • Insert statistical analyses including correlations, geographic associations, etc. See Chapter 3, textbox 3-2. • You may want to look at other types of evidence that can be used for elimination in Unit II, Part B and C.

If you feel that you have enough case specific data to eliminate some causes, proceed to Unit III Step 1 (Eliminate Alternatives). If not, proceed to Unit II Part B.

UNIT II

Part B. Measurements associated with the causal mechanism (Chapter 3.3).

- Evidence can be used for Elimination (**E**) Diagnosis (**D**) or Strength of Evidence (**S**), as noted below.
- Prepare a separate table for each candidate cause.
- Use this as a reminder of types of data that could be used in the analysis. Not all questions may be appropriate.

Candidate Cause: _______________________

Example Questions:	Yes/No/ Question Not Relevant	Associated Causal Characterization Method in Unit III*	Supporting Analysis
Are symptoms or other responses specific to or characteristic of a type of stressor found in organisms from the impaired community?		**D, S**	
Are there internal measures of exposure (e.g., body burdens, biomarkers) found in organisms from the impaired community?		**E, D, S**	
Is an intermediate product of an ecological process present?		**E, S**	
Do distributions of stressors and receptors coincide?		**E, S**	

Example Questions:	Yes/No/ Question Not Relevant	Associated Causal Characterization Method in Unit III*	Supporting Analysis
Have there been expected changes in the abundance of predators, prey, or competitors?		**S**	
Are there expected effects on other receptors?		**S**	
Other			

***E = Elimination; D = Diagnosis; S = Strength of Evidence**

If you feel that your evidence can be used to identify the cause through diagnosis, go to Unit III, Step 2. If not, continue with the analysis of evidence in Unit II Parts C and D.

UNIT II

Part C. Associations of effects mitigation with manipulation of causes (Chapter 3.4).

- Evidence can be used for elimination ONLY if it is from the site.
- Prepare a separate table for each candidate cause.
- Use this as a reminder of the types of data that could be used in the analysis. Not all questions may be appropriate.

Candidate Cause: ________________________

Questions:	**Yes/No/ Information not available/ Question not Applicable**	**Asso-ciated Causal Charac-teriza-tion Method in Unit III***	**Supporting Analysis**
Does elimination of the source reduce or eliminate the effect?		**S, E**	
Does the introduction of previously unexposed organisms result in an effect?		**S**	
Does the isolation of organisms from one cause reveal the effects of others?		**S**	

Questions:	Yes/No/ Information not available/ Question not Applicable	Asso-ciated Causal Charac-teriza-tion Method in Unit III*	Supporting Analysis
Does the testing of chemical fractions of site media result in toxicity being associated with a particular fraction (i.e., TIE)?		**S**	
Other			

***E = Elimination; D = Diagnosis; S = Strength of Evidence**

If you have enough data to determine the cause, proceed to Unit III Step 1 (Elimination) or Step 2 (Diagnosis) or Step 3 (Strength of Evidence), as appropriate. If not or uncertain, proceed to Unit II Part D.

UNIT II

Part D. Using effects data from elsewhere (Chapter 3.2).

- Use this table to incorporate data from other situations that support the analysis. Not all questions may be appropriate for a given candidate cause.
- This evidence is applicable to Strength of Evidence **(S)** characterization method.
- Prepare a separate table for each candidate cause.

Candidate Cause: ______________________

Type of Candidate Cause	Characterization of Exposure (Intensity, Time, and Space)	Data Available? Yes (note location of data)/No	Exposure-Response (E-R) Relationship	E-R Available? Yes (note location of data) /No/Not Relevant	Would effects be expected at the environmental conditions seen in the case? (Yes/No)	Location of supporting analysis
Chemical	What is the concentration in the medium at the site?		What is the concentration-response relationship (seen in the lab or the field)?			
	What is the internal concentration in organisms at the site?		What is the internal external concentration-response relationship (seen in the lab or the field)?			
	What is the concentration in the biomarker at the site?		What is the biomarker-response relationship?			

Type of Candidate Cause	Characterization of Exposure (Intensity, Time, and Space)	Data Available? Yes (note location of data)/No	Exposure-Response (E-R) Relationship	E-R Available? Yes (note location of data) /No/Not Relevant	Would effects be expected at the environmental conditions seen in the case? (Yes/No)	Location of supporting analysis
Effluent	What is the dilution of the effluent at the location of the impairment?		What are the laboratory test (i.e., WET) results from 100% effluent or diluted effluent?			
Contaminated ambient media	What were the location and time of collection and the results of analyses?		What are the results of laboratory tests of ambient media?			
Habitat	What are the structural attributes of the habitat?		Are empirical models available that relate habitat characteristics to biological responses ?			
Water Withdrawal or Drought	Are hydrograph readings and summary statistics (e.g., 7Q10) available?		What are the results of instream flow models (e.g., IFIM)?			
Thermal Energy	Are temperature records available?		What are the thermal tolerances of the impacted organisms?			

Type of Candidate Cause	Characterization of Exposure (Intensity, Time, and Space)	Data Available? Yes (note location of data)/No	Exposure-Response (E-R) Relationship	E-R Available? Yes (note location of data) /No/Not Relevant	Would effects be expected at the environmental conditions seen in the case? (Yes/No)	Location of supporting analysis
Siltation (Suspended)	What is the total suspended solids (TSS) concentration?		What is the concentration-response relationship (seen in the lab or field)?			
Siltation (Bed-load)	What is the degree of embeddedness and texture of the silt?		Are empirical models available to characterize the effects?			
Dissolved Oxygen and Oxygen-Demanding Contaminants (e.g., BOD, COD)	Review the dissolved oxygen data (esp. predawn).		What is the concentration-response relationship (from lab or other field studies)?			
	Review the BOD, COD data from the source.		Are there oxygen demand models that can be used to predict effects?			
Excess Mineral Nutrients	What were the dissolved mineral nutrient concentrations?		What is the concentration-response relationship (from lab or other field studies)?			
			Are there nutrient/eutrophication models that can be used to predict effects?			

Type of Candidate Cause	Characterization of Exposure (Intensity, Time, and Space)	Data Available? Yes (note location of data)/No	Exposure-Response (E-R) Relationship	E-R Available? Yes (note location of data) /No/Not Relevant	Would effects be expected at the environmental conditions seen in the case? (Yes/No)	Location of supporting analysis
Nonindigenous Species	Is a nonindigenous species present or abundant?		Are ecological models available to characterize the effects?			
Pathogen	Is a pathogen present? If so, is it abundant?		Are any symptoms or diseases observed?			
Other						

Go to Unit III, Characterize Causes.

UNIT III. CHARACTERIZE CAUSES

Step 1. Eliminate Alternatives (Section 4.1.1) ***and compare supporting evidence where causes were eliminated.***

- For each candidate cause indicate Yes, No, No Evidence (NE), or Not Applicable (NA).
- If more than one stressor is necessary for a cause to be sufficient (i.e., temperature and dissolved oxygen), indicate response for each stressor.
- Use extra pages for more than 3 candidate causes.
- Provide comments as necessary.

Case-Specific Consideration	**Candidate Cause # 1** **(Yes / No / NE / NA)**	**Candidate Cause # 2** **(Yes / No / NE / NA)**	**Candidate Cause # 3** **(Yes / No / NE / NA)**
Temporal Co-occurrence Did the effect precede the stressor in time? *(If the effects preceded a proposed cause and effects are not obscured by another sufficient cause, then it cannot be the primary cause.)*			
Temporal Gradient Did the effect increase or decrease over time in association with an increase or decrease in the stressor? *(If the effect increases or decreases over time without a corresponding increase or decrease in the stressor, then the stressor cannot be the primary cause.)*			

Case-Specific Consideration	**Candidate Cause # 1** **(Yes / No / NE / NA)**	**Candidate Cause # 2** **(Yes / No / NE / NA)**	**Candidate Cause # 3** **(Yes / No / NE / NA)**
Spatial Co-occurrence Is there an upstream/downstream conjunction of candidate cause and effect? *(If the effect occurs upstream of the source or does not occur regularly downstream, e.g., is distributed spatially independently of a plume, sediment deposition areas, etc., and effects are not obscured by another sufficient cause, then the candidate cannot be the primary cause).*			
Co-occurrence with Reference Site(s) Is there a reference site/impaired site conjunction of candidate cause and effect? *(If the cause occurs at reference sites as well as the impaired sit, it can be eliminated.)*			
Spatial Gradient Does the effect increase or decrease across a given region in association with an increase or decrease in the stressor? *(If the effect increases or decreases over a given region without a corresponding increase or decrease in the stressor, then the stressor cannot be the primary cause.)*			

Case-Specific Consideration	**<u>Candidate Cause # 1</u>** **(Yes / No / NE / NA)**	**<u>Candidate Cause # 2</u>** **(Yes / No / NE / NA)**	**<u>Candidate Cause # 3</u>** **(Yes / No / NE / NA)**
Biological Gradient Is a decrease in the magnitude or proportion of an effect seen along a decreasing gradient of the stressor? *(A constant or increasing level of effect with decreasing exposure would eliminate a cause.)*			
Complete Exposure Pathway, **Question 1:** Is there evidence that the stressor did not co-occur with, contact, or enter the receptor(s) showing the effect? *(If there is no route of exposure, or, for appropriate stressors, if tissue burdens or other measures of exposure were not found to occur in affected organisms, the cause may be eliminated.)*			
Complete Exposure Pathway, **Question 2:** Is there evidence that a necessary intermediate step in the causal chain of events did not occur? *(If a link in a known chain of events can be shown to be missing, the cause may be eliminated.)*			

Case-Specific Consideration	**Candidate Cause # 1** **(Yes / No / NE / NA)**	**Candidate Cause # 2** **(Yes / No / NE / NA)**	**Candidate Cause # 3** **(Yes / No / NE / NA)**
Experiment, Temporality Did the effects continue when the candidate cause was removed (allowing for rates of recovery)? *(If effects continue despite elimination of the candidate cause, that cause can be eliminated.)*			
Other			

After completing Step 1 (above) for each candidate cause listed in Unit I:

- If only **one** candidate cause remains, elimination is definitive. Go to Unit IV.
- If **more than one** candidate cause remains, go back to Unit II, Part B. If Unit II Part B is complete, go to Unit III Step 2.
- If **no** candidate causes remains, go to Unit V.

UNIT III

Step 2. Characterize cause using diagnostic evidence (Section 4.1.2).

- If diagnostic evidence was found in Unit II Part D, determine if the evidence is sufficient to define the cause using this table.
- If evidence is not sufficient to diagnose the cause, it may still be used in the strength of evidence in Unit III Step 3.
- Use extra pages for more than 3 candidate causes.

Candidate Cause	Type of Diagnostic Evidence	Description of Evidence
# 1		
# 2		
# 3		

After completing Step 2 for all causes remaining after the elimination step (Step 1):

- If diagnosis is definitive. Go to Unit IV.
- If **diagnosis is uncertain**, go back to Unit II Parts B, C and D. **If Unit II Parts B, C, and D are complete, proceed to Unit III Step 3.**

UNIT III

Step 3. Analyze strength of evidence (Section 4.1.3) *for Case-Specific Considerations.*

- Use extra pages for more than 3 candidate causes.

Causal Considerations and possible scores	**Candidate Cause # 1**		**Candidate Cause # 2**		**Candidate Cause # 3**	
	Evidence and Literature Citation	**Score**	**Evidence and Literature Citation**	**Score**	**Evidence and Literature Citation**	**Score**
Co-occurrence Compatible (+), Uncertain (0), Incompatible (- - -), No evidence (NE) *(The stressor has either contacted the affected organisms, their food source, or some parameter that can affect the organisms.)*						

Causal Considerations and possible scores	Candidate Cause # 1		Candidate Cause # 2		Candidate Cause # 3	
	Evidence and Literature Citation	**Score**	**Evidence and Literature Citation**	**Score**	**Evidence and Literature Citation**	**Score**
Temporality Compatible (+), Uncertain (0), Incompatible (- - -), No evidence (NE) *(A cause must always precede its effects.)*						
Consistency of Association Invariant (++), In many places and times (+), At background frequencies (-), No Evidence (NE) *(The repeated observation of a similar relationship of the effect and candidate cause in different places and times.)*						

Causal Considerations and possible scores	**Candidate Cause # 1**		**Candidate Cause # 2**		**Candidate Cause # 3**	
	Evidence and Literature Citation	**Score**	**Evidence and Literature Citation**	**Score**	**Evidence and Literature Citation**	**Score**
Biological Gradient Strong and monotonic (+++), Weak or other than monotonic (+), None (-), Clear association but wrong sign (- - -), Not applicable (NA) *(The effect increases in a regular manner with increasing exposure.)*						

Causal Considerations and possible scores	**Candidate Cause # 1**		**Candidate Cause # 2**		**Candidate Cause # 3**	
	Evidence and Literature Citation	**Score**	**Evidence and Literature Citation**	**Score**	**Evidence and Literature Citation**	**Score**
Complete Exposure Pathway Evidence for all steps (++), Incomplete evidence (+), Ambiguous (0), Some steps missing or implausible (-), No evidence (NE) *(The stressor co-occurs with or contacts the receptor(s).)*						

Causal Considerations and possible scores	**Candidate Cause # 1**		**Candidate Cause # 2**		**Candidate Cause # 3**	
	Evidence and Literature Citation	**Score**	**Evidence and Literature Citation**	**Score**	**Evidence and Literature Citation**	**Score**
Experiment Experimental studies Concordant (+++), Ambiguous (0), Inconcordant (- - -) No evidence (NE) *(Toxicity tests or other controlled experimental studies demonstrated that the candidate cause can induce the observed effect.)*						

UNIT III

Step 4. Analyze strength of evidence *(Section 4.1.3) using Evidence from Other Situations or from Biological Knowledge.*

- Use extra pages for more than 3 candidate causes.

Causal Consideration and possible scores	**Candidate Cause # 1**		**Candidate Cause # 2**		**Candidate Cause # 3**	
	Evidence and Literature Citation	**Score**	**Evidence and Literature Citation**	**Score**	**Evidence and Literature Citation**	**Score**
Plausibility: Mechanism Evidence of Mechanism (++), Plausible (+), Not Known (0), Implausible (-) *(It is plausible that the effect resulted from the cause given what is known about the biology, physics, and chemistry of the candidate cause, the receiving environment, and the affected organisms.)*						

Causal Consideration and possible scores	**Candidate Cause # 1**		**Candidate Cause # 2**		**Candidate Cause # 3**	
	Evidence and Literature Citation	**Score**	**Evidence and Literature Citation**	**Score**	**Evidence and Literature Citation**	**Score**
Plausibility: Stressor-Response Quantitatively consistent (+++), Concordant (+), Ambiguous (0), Inconcordant (-), No evidence (NE) *(Given a known relationship between the candidate cause and the effect, effects would be expected at the level of stressor seen in the environment.)*						

Causal Consideration and possible scores	**Candidate Cause # 1**		**Candidate Cause # 2**		**Candidate Cause # 3**	
	Evidence and Literature Citation	**Score**	**Evidence and Literature Citation**	**Score**	**Evidence and Literature Citation**	**Score**
Consistency of Association Invariant (+++), In most places (++), In some places (+), At background frequency (-), Not applicable (NA) *(The repeated observation of the effect and candidate cause is similar in different places and times.)*						

Causal Consideration and possible scores	**Candidate Cause # 1**		**Candidate Cause # 2**		**Candidate Cause # 3**	
	Evidence and Literature Citation	**Score**	**Evidence and Literature Citation**	**Score**	**Evidence and Literature Citation**	**Score**
Analogy: Positive Analogous cases: Many or few but clear (++), Few or unclear (+), None (0) *(The hypothesized relationship between cause and effect similar to other well-established cases.)*						
Analogy: Negative Analogous cases: Many or few but clear (- -), Few or unclear (-), None (0) *(The hypothesized relationship between cause and effect is dissimilar to other well-established cases.)*						

Causal Consideration and possible scores	Candidate Cause # 1		Candidate Cause # 2		Candidate Cause # 3	
	Evidence and Literature Citation	**Score**	**Evidence and Literature Citation**	**Score**	**Evidence and Literature Citation**	**Score**
Specificity of Cause* ***Note: only applicable if the cause is plausible or is consistently associated with the effect.*** Only possible cause (+++), One of a few (+), One of many (0), Not applicable (NA) *(The effect observed at the site is known to have only one or a few known causes.)*						

Causal Consideration and possible scores	Candidate Cause # 1		Candidate Cause # 2		Candidate Cause # 3	
	Evidence and Literature Citation	**Score**	**Evidence and Literature Citation**	**Score**	**Evidence and Literature Citation**	**Score**
Experiment Experimental studies: Concordant (+++), Ambiguous (0), Inconcordant (- - -), No evidence (NE) *(Toxicity tests or other controlled experimental studies demonstrated that the candidate cause can induce the observed effect.)*						

Causal Consideration and possible scores	Candidate Cause # 1		Candidate Cause # 2		Candidate Cause # 3	
	Evidence and Literature Citation	**Score**	**Evidence and Literature Citation**	**Score**	**Evidence and Literature Citation**	**Score**
Predictive Performance Prediction: Confirmed specific or multiple (+++), Confirmed general (++), Ambiguous (0), Failed (- - -), No evidence (NE) *(The candidate cause has any initially unobserved properties that were predicted to occur and the prediction was subsequently confirmed at the site.)*						

UNIT III

Step 5. Analyze strength of evidence *(Section 4.1.3) based on multiple lines of evidence.*

- Use extra pages for more than 3 candidate causes.

Causal Consideration and possible scores	Candidate Cause # 1		Candidate Cause # 2		Candidate Cause # 3	
	Evidence and Literature Citation	Score	Evidence and Literature Citation	Score	Evidence and Literature Citation	Score
Consistency of Evidence All consistent (+++), Most consistent (+), Multiple inconsistencies (- - -) *(The hypothesized relationship between the cause and effect is consistent across all available evidence.)*						

Causal Consideration and possible scores	**Candidate Cause # 1**		**Candidate Cause # 2**		**Candidate Cause # 3**	
	Evidence and Literature Citation	**Score**	**Evidence and Literature Citation**	**Score**	**Evidence and Literature Citation**	**Score**
Coherence of Evidence Evidence: Inconsistency explained by a credible mechanism (+), No known explanation (0) **No entry if all consistent** *(A mechanistic conceptual model explains any apparent inconsistencies among the lines of evidence.)*						

Compare evidence among the candidate causes, then go to Unit IV to summarize your findings.

IV. SUFFICIENCY OF EVIDENCE (Chapter 4.2)

Most Likely Candidate Cause: ____

Is Evidence Sufficient for the Management Purpose?

☐ YES SI COMPLETE, REPORT RESULTS

☐ NO GO TO UNIT V, RECONSIDER IMPAIRMENT

Summary of Characterization		
Candidate Cause	**Cause**	**Reasoning & Confidence**
# 1.		
# 2.		
# 3.		

V. RECONSIDER IMPAIRMENT

Does Biological Impairment Really Exist?
(Section 5.1)

Reconsider the impairment by auditing the quality of the methods used to generate and manage the data, by using better analysis tools, and by eliminating any suspicious data or analyses.
Describe Reconsideration:

____

____

____

____

Were effects real?

- ☐ **NO** **SI COMPLETE, REPORT RESULTS.**
- ☐ **YES** **GO TO UNIT VI, COLLECT MORE INFORMATION.**

VI. COLLECT MORE INFORMATION (Section 5.2)

Were all reasonable causes analyzed?

- ☐ **NO** **Go to Follow-on 1.**
- ☐ **YES** **Go to Follow-on 2.**

Follow-on 1: **Make sure that all reasonable causes were analyzed.**

- **If additional scenarios are indicated, repeat process, beginning at Unit 1.**
- **If a good faith effort was implemented with reasonable time and resource expenditures, consult management goals and determine if the process should be ended with inconclusive results.**

SI COMPLETED, REPORT RESULTS AS INCONCLUSIVE.

Follow-on 2: **Look at the supporting evidence in Unit II, Analyze Evidence.**

- **Prioritize information needs for likely candidate causes, collect new information and repeat the process, beginning at Unit 1.**
- **If a good faith effort was implemented with reasonable time and resource expenditures, consult management goals and determine if the process should be ended with inconclusive results.**

SI COMPLETED, REPORT RESULTS AS INCONCLUSIVE.

APPENDIX C
GLOSSARY OF TERMS

Appendix C

Glossary of Terms

Ambient monitoring: All forms of monitoring conducted beyond the immediate influence of a discharge pipe or injection well and may include sampling of sediments and living resources.

Ambient waters: water bodies that are in the environment.

Analogy: a comparison of two things, based on their similarity in one or more respects. In SI, the criterion of an analogy refers specifically to similar causes.

Bioassessment (biological assessment): evaluation of the condition of an ecosystem that uses biological surveys and other direct measurements of the resident biota.

Biocriteria (biological criteria): numerical values or narrative expressions that describe the reference biological condition of aquatic communities inhabiting waters of a given designated aquatic life use. Biocriteria are benchmarks for evaluation and management of water resources

Biogenic: produced by biological processes. For example, organic acids produced by decomposition of plant litter are biogenic acids.

Biological gradient: a regular increase or decrease in a measured biological attribute with respect to space (e.g., below an outfall), time (e.g., since a flood), or an environmental property (e.g., temperature).

Biomarker: contaminant-induced physiological, biochemical, or histological response of an organism.

Body burden: the concentration of a contaminant in a whole organism or a specified organ or tissue.

Candidate cause: a hypothesized cause of an environmental impairment which is sufficiently credible to be analyzed.

Categorical regression: regression analysis in which the dependent variable is defined by a categorical scale rather than as a count or continuous variable.

Causal analysis: a process in which data and other information are organized and evaluated using quantitative and logical techniques to determine the likely cause of an observed condition.

Causal mechanism: the process by which a cause induces an effect.

Causal relationship: the relationship between a cause and its effect.

Causal association: a correlation or other association between measures or observations of two entities or processes which occurs because of an underlying causal relationship.

Causal evidence: the results of an analysis of data to reveal an association between the environmental condition and a candidate cause.

Causal inference: the component of a causal analysis that is specifically concerned with the interpretation of the evidence to determine the most likely cause.

Causal characterization: a step in the stressor identification process in which the proposed cause is described, the evidence for its causal relationship to the impairment is summarized, and uncertainties are presented.

Causal considerations: logical categories of evidence that are consistently applied to support or refute a hypothesized cause. A causal consideration (e.g., biological gradient) is evaluated using causal evidence (e.g., a regression of benthic invertebrate diversity against sediment PCB concentration).

Cause: 1. that which produces an effect (a general definition).
2. a stressor or set of stressors that occur at an intensity, duration and frequency of exposure that results in a change in the ecological condition (a SI-specific definition).

Co-occurrence: the spatial co-location of the candidate cause and effect.

Coherency of evidence: the final consideration in a strength of evidence analysis. If the results of all of the causal considerations in a strength of evidence analysis are not consistent, they may still be coherent, if a mechanistic conceptual or mathematical model explains the apparent inconsistencies.

Complete exposure pathway: the physical course a stressor takes from the source to the receptors (e.g., organisms or community) of interest. (Evidence for a complete exposure pathway is case-specific and may include measurements such as body burdens of chemicals, presence of parasites or pathogens, or biomarkers of exposure.)

Concentration-response model: a quantitative (usually statistical) model of the relationship between the concentration of a chemical to which a population or community of organisms is exposed and the frequency or magnitude of a biological response.

Consideration: see Causal consideration.

Consistency of association: the degree to which an effect and candidate cause have been determined to co-occur in different places or times.

Consistency of evidence: the degree to which the causal considerations in a strength of evidence analysis are in agreement concerning a candidate cause.

Diagnostic analysis: a type of causal analysis in which effects that are characteristic of a particular cause are used to determine whether that candidate cause may be responsible for an impairment.

Diagnostic protocol: a standard procedure for performing a diagnostic analysis.

Dilution ratio: the ratio of the stream flow to the wastewater flow

Ecoepidemiology: the study of the nature and causes of effects on ecological systems.

Endpoint species: a species that is the object of an assessment or test.

Eutrophication: enrichment of a water body with nutrients, resulting in high levels of primary production, often leading to depletion of dissolved oxygen.

Experiment: the manipulation of a candidate cause by eliminating a source or altering exposure so as to evaluate its relationship to an effect.

Expert judgement: a method of causal inference based on the knowledge and skill of the assessors rather than a formal method.

Exposure: the co-occurrence or contact of a stressor and the resource that becomes impaired.

Exposure-response relationships:	a qualitative or quantitative (usually statistical) model of the relationship between an exposure metric (e.g., the concentration of a chemical or the abundance or an exotic species) to which a population or community of organisms is exposed and the frequency or magnitude of a biological response.
Impairment:	a detrimental effect on the biological integrity of a water body that prevents attainment of the designated use.
Indirect causation:	the induction of effects through a series of cause-effect relationships, so that the impaired resource may not even be exposed to the initial cause.
Indirect effects:	changes in a resource that are due to a series of cause-effect relationships rather than to direct exposure to a contaminant or other stressor.
Inferential logic:	a process for reasoning from the evidence to a necessary and specific conclusion.
Initial response:	the response of an organism, population or community to direct exposure to a stressor.
Intermediate processes:	processes that occur between the occurrence of a stressor in an ecosystem and the induction of the effect of concern. For example, the reduction in algal abundance is an intermediate process between the introduction of a non-native filter feeder and the reduction in abundance of native planktivorous species.
Internal exposure:	exposure of an organism to bioaccumulated contaminants.
Logic of abduction:	inference from data to the hypothesis that best accounts for the data.
Mechanism:	the process by which a system is changed.
Necropsy:	a post-mortem examination or inspection intended to determine the cause of death or the nature of pathological changes.
Negative evidence:	evidence that tends to refute a candidate cause.
Opportunistic:	having the ability to exploit newly available habitats or resources.
Pathogens:	organisms that are capable of inducing a disease in a susceptible host.

Plausibility: the degree to which a cause and effect relationship would be expected, given known facts.

Positive evidence: evidence that tends to support a candidate cause.

Predictive performance: the degree to which a candidate cause has led to predictions concerning conditions in the receiving system which have been subsequently confirmed by observation or measurement.

Principal cause: the cause that makes the largest contribution to the effect.

Pseudoreplication: the treatment of multiple samples that are subject to the same treatment as replicates for statistical purposes. For example, multiple samples of benthic invertebrates taken in a channelized stream are pseudo- replicates because they are not independent. True replicates would be taken from different channelized streams.

Publicly Owned Treatment Works (POTW): a water treatment facility, as defined by Section 212 of the Clean Water Act, that is used in the storage, treatment, recycling, and reclamation of municipal sewage or industrial wastes of a liquid nature and is owned by a municipality or other governmental entity. It usually refers to sewage treatment plants.

Receptors: organisms, populations, or ecosystems that are exposed to a contaminant or other stressor.

Replicate: (a) one of a set of independent systems which have been randomly assigned a treatment; or (b) to generate a set of such systems.

Source: an origination point, area, or entity that releases or emits a stressor. A source can alter the normal intensity, frequency, or duration of a natural attribute, whereby the attribute then becomes a stressor.

Spatial gradient: a graded change in the magnitude of some quantity or dimension measured on a transect

Specificity: the quality of being specific rather than general.

Specificity of cause: only one candidate cause or a few similar causes can induce the observed effect.

Specificity of effect: one type of effect is characteristically induced by a candidate cause. The absence of that effect is evidence for eliminating the candidate cause.

Strength-of-evidence analysis: an inferential process that uses all relevant evidence in a systematic process to determine which candidate cause is most likely to have induced the effect of concern.

Strength of association: the size of the effect produced by an increment in the candidate cause. A candidate cause that is associated with a large change in the level of effect is more likely to be the true cause than one that is weakly associated.

Stressor: any physical, chemical, or biological entity that can induce an adverse response.

Supplemental Environmental Project (SEP): a special program that is often used to grant injunctive relief.

Symptomatology: a set of signs of the action of a causal agent on organisms. A set of symptoms with a common cause constitutes a symptomatology.

Temporal relationship: the relationship between the time of occurrence of a candidate cause and of the effect of concern.

Temporal gradient: a graded change in the magnitude of some quantity or dimension measured over time.

Total Maximum Daily Load (TMDL): the total allowable pollutant load to a receiving water such that any additional loading will produce a violation of water-quality standards.

Toxicity Reduction Evaluation (TRE): a site-specific study conducted in a stepwise process designed to identify the causative agent(s) of effluent toxicity, isolate the sources of toxicity, evaluate the effectiveness of toxicity control options, and then confirm the reduction in effluent toxicity.

Toxicity Identification and Evaluation (TIE): a process that identifies the toxic components of an effluent or ambient medium by a process of chemically manipulating the effluent or medium and testing the resulting material.

APPENDIX D
LITERATURE CITED

Appendix D

Literature Cited

Albers, P. 1995. Petroleum and individual polycyclic aromatic hydrocarbons, Pages 330-355 *in* Hoffman, David J. et al, ed. *Handbook of Ecotoxicology*, CRC Press, Boca Raton, Florida.

Adams, D.F. 1963. Recognition of the effects of fluorides on vegetation. *J. Air Pollut. Control Assoc*. 13:360-362.

Allan, J.D. 1995. *Stream ecology: structure and function of running waters*. Chapman and Hall Publishers, London.

Baumann, P.C., and J.C. Harshbarger. 1995. Decline in liver neoplasms in wild brown bullhead catfish after coking plant closes and environmental PAHs plummet. *Environ. Health Perspect.* 103:168-170.

Baumann, P.C., I.R. Smith, and C.D. Metcalfe. 1996. Linkage between chemical contaminants and tumors in benthic Great Lakes fish. *J. Great Lakes Res.* 22(2):131-152.

Beyers, D.W. 1998. Causal inference in environmental impact studies. *J. North Amer. Benthol. Soc.* 17: 367-373.

Botti, C., P. Comba, F. Forastiere, and L. Settimi. 1996. Causal inference in environmental epidemiology: the role of implicit values. *Sci. Total Environ.* 184:97-101.

Blus, L.J., and C.F. Henny. 1997. Field studies on pesticides and birds: unexpected and unique relations. *Ecological Applications*. 7:1125-1132.

Carpenter, S.R., N.F. Caraco, D.L. Correll, R.W. Howarth, A.N. Sharpley, and V.H. Smith. 1988. Non-point pollution of surface waters with phosphorus and nitrogen. *Ecological Applications* 8(3):559-568.

Clemen, R.T. 1986. *Making hard decisions. An introduction to decision analysis*. 2nd ed. Duxbury Press, Belmont, CA.

Clements, W.H. 1994. Benthic invertebrate community responses to heavy metals in the Upper Arkansas River Basin, Colorado. *J. N. Am. Benthol. Soc.* 13:30-44.

Clements, W.H., and P.M. Kiffney. 1994. Integrated laboratory and field approach for assessing impacts of heavy metals at the Arkansas River, Colorado. *Environmental Toxicology and Chemistry* 13:397-404.

Clements, W.H., D.M. Carlisle, J.M. Lazorchak, and P.C. Johnson. 2000. Heavy metals structure benthic communities in Colorado mountain streams. *Ecol. Appl.* 10:626-638.

Cormier, S.M., E.L.C. Lin, F.A. Fulk, and B. Subramanian. 2000a. Estimation of exposure criteria values for biliary polycyclic aromatic hydrocarbon metabolite concentration in white suckers (*Catostomus commersoni*). *Environmental Toxicology and Chemistry* 19:1120-1126.

Cormier, S.M., E.L.C. Lin, M.R. Millward, M.K. Schubauer-Berigan, D. Williams, B. Subramanian, R. Sanders, B. Counts, and D. Altfater. 2000b. Using regional exposure criteria and upstream reference data to characterize spatial and temporal exposures to chemical contaminants. *Environmental Toxicology and Chemistry* 19:1127-1135.

Cormier, S.M., M. Smith, S. Norton, and T. Neiheisel. 2000c. Assessing ecological risk in watersheds: a case study of problem formulation in the Big Darby Creek Watershed, Ohio, USA. *Env. Tox. and Chemistry* Vol. 19, No. 4(2) 1082-1096.

Courtemanch, D.L., P. Mitnik, and L. Tsomides. 1997. Dec. 8, Memorandum to Greg Wood, Maine Department of Environmental Protection Licensing Section, Augusta, Maine.

Davenport, T.E., and M.H. Kelly. 1983. *Water resource data and preliminary trend analysis for the Highland Silver Lake Monitoring and Evaluation Project, Madison County, Illinois. Phase II.* Report No. IEPA/WPC/83-013. Illinois Environmental Protection Agency, Springfield.

Davies, S.P., L. Tsomides, D.L. Courtemanch, and F. Drummond. 1995. *Maine biological monitoring and biocriteria development program.* Maine Department of Environmental Protection, Augusta, Maine.

Davies, S.P., and L. Tsomides. 1997. *Methods for biological sampling and analysis of Maine's inland waters.* DEP-LW/07-A97. Maine Department of Environmental Protection, Augusta, Maine.

Davies, S.P., L. Tsomides, J.L. DiFranco, and D.L. Courtemanch. 1999. *Biomonitoring retrospective: Fifteen year summary for Maine rivers and streams.* DEPLW1999-26. Maine Department of Environmental Protection, Augusta, Maine.

Dodds, K., and E.B. Welsh. 2000. Establishing nutrient criteria in streams. *The North American Benthological Society* 19:186-196.

Dourson, M.L., L.K. Teuschler, P.R. Durkin, and W.M. Stiteler. 1997. Categorical regression of toxicity data: A case study using aldicarb. *Reg. Toxicol. Pharmacol.* 25:121-129.

Edwards, R., and J. Riepenhoff. 1998. State turns to feds for cleanup. *Columbus Dispatch*, April 28, 1998.

Edwards, A.C., H. Twist, and G.A. Codd. 2000. Assessing the impact of terrestrially derived phosphorus on flowing water systems. *Journal of Environmental Quality.* 29:117-124.

Edwards, C.J., B.L. Griswold, R.A. Tubb, E.C. Weber, and L.C. Woods. 1984. Mitigating effects of artificial riffles and pools on the fauna of a channelized warmwater stream. *North American Journal of Fisheries Management* 4:194-203.

Eisler, R. 2000a. Polycyclic aromatic hydrocarbons. Pages 1343-1411 *in Handbook of Chemical Risk Assessment.* Vol. II. Lewis Publishers, Boca Raton, FL.

_______. 2000b. *Handbook of Chemical Risk Assessment*. Vol. I. Lewis Publishers, Boca Raton, FL.

Fox, G.A. 1991. Practical causal inference for ecoepidemiologists. *J. Toxicol. Environ. Health* 33:359-373.

Gilbertson, M., T. Kubiak, J. Ludwig, and G. Fox. 1991. Great Lakes embryo mortality, edema, and deformities syndrome (GLEMEDS) in colonial fish-eating birds: similarity to chick-edema disease. *J. Toxicology and Environmental Health* 33:455-520.

Gmur, D.J., and U. Varanasi. 1982. Characterization of benzo[a]pyrene metabolites isolated from muscle, liver, and bile of a juvenile flatfish. *Carcinogenesis* 5:1397-1403.

Grier, J.W. 1982. Ban of DDT and subsequent recovery of reproduction in bald eagles. *Science* 218:1232-1234.

Hackney, J.D., and W.S. Kinn. 1979. Koch's postulates updated: A potentially useful application to laboratory research and policy analysis in environmental toxicology. *Amer. Rev. Respir. Dis.* 1119:849-852.

Hickey, C.W., and W.H. Clements. 1998. Effects of heavy metals on benthic macroinvertebrate communities in New Zealand streams. *Environmental Toxicology and Chemistry* 17:2338-2346.

Hilborn, R., and M. Mangel. 1997. *The ecological detective: confronting models with data.* Princeton U. Press, Princeton, NJ.

Hill, A.B. 1965. The environment and disease: Association or causation. *Proceed. Royal Soc. Medicine* 58:295-300.

Hilsenhoff, W.L. 1987. An improved biotic index of organic stream pollution. *Great Lakes Entomol.* 20:31-39.

Huggins, D.G., and M.F. Moffett. 1988. *Proposed biotic and habitat indices for use in Kansas streams.* Report No. 35, Kansas Biological Survey, Lawrence.

Hurlbert, S.H. 1984. Pseudoreplication and the design of ecological field experiments. *Ecological Mono.* 54:187-211.

Josephson, J.R., and S.G. Josephson. 1996. *Abductive Inference.* Cambridge University Press, Cambridge.

Jorgensen, S.E. 1994. *Fundamentals of ecological modelling.* Elsevier, Amsterdam.

Kansas Department of Health and Environment (KDHE). 1993. *Kansas list of water quality limited surface waters (303(d) list).* Kansas Department of Health and Environment, Division of Environment, Office of Science and Support, Topeka., KS.

__________. 1998. *Kansas water quality limited segments (303(d) list).* Kansas Department of Health and Environment, Division of Environment, Bureau of Environmental Field Services, Topeka, KS.

__________. 2000. *Kansas water quality assessment (305(b) report).* Kansas Department of Health and Environment, Division of Environment, Bureau of Environmental Field Services, Topeka, KS.

Karr, J.R., and I.J. Schlosser. 1977. *Impact of nearstream vegetation and stream morphology on water quality and stream biota.* EPA-600-3-77-097. U.S. Environmental Protection Agency, Environmental Research Laboratory, Athens, GA.

Kiffney, P.M., and W.H. Clements. 1994a. Structural responses of benthic macroinvertebrate communities from different stream orders to zinc. *Environmental Toxicology and Chemistry* 13:389-395.

__________. 1994b. Effects of heavy metals on a macroinvertebrate assemblage from a Rocky Mountain stream in experimental microcosms. *J. N. Am. Benthol. Soc.* 13:511-523.

Lehman, J.T. 1986. Control of eutrophication in Lake Washington: case study. Pages 301-316 *in Ecological knowledge and environmental problem-solving: Concepts and case studies*. National Academy Press, Washington, D.C.

Lin, E.L.C., S.M. Cormier, and J.A. Torsella. 1996. Fish biliary polycyclic aromatic hydrocarbon metabolites estimated by fixed-wavelength fluorescence: comparison with HPLC-fluorescent detection. *Ecotoxicol. Environ. Safety* 35:16-23.

Lin, E.L.C., T.W. Neiheisel, B. Subramanian, D.E. Williams, M.R. Millward, and S.M. Cormier. Historical monitoring of biomarkers of exposure of brown bullhead in the remediated Black River, Ohio and two other Lake Erie tributaries. Submitted to *Journal of Great Lakes Research.*

Long, E.R., L.J. Field, and D.D. MacDonald. 1998. Predicting toxicity in marine sediments with numerical sediment quality guidelines. *Envir. Toxicol. Chem.* 17:714-727.

Meyer, F.P., and L.A. Barclay. 1990. *Field manual for the investigation of fish kills.* Resource Pub. 177. U.S. Fish and Wildlife Service, Washington, D.C.

Miltner, R.J., and E.T. Rankin. 1998. Primary nutrients and the biotic integrity of rivers and streams. *Freshwater Biology* 40:145-158.

Mitnik, P. 1994. *Presumpscot River waste load allocation.* Maine Department of Environmental Protection, Augusta, Maine.

________. 1998. *Presumpscot River supplemental report to waste load allocation.* Maine Department of Environmental Protection, Augusta, Maine.

Nelson S.M., and R.A. Roline. 1996. Recovery of a stream macroinvertebrate community from mine drainage disturbance. *Hydrobiologia* 339:73-84.

Norberg-King, T., and D. Mount. 1986. *Validity of effluent and ambient toxicity tests for predicting biological impact*. Skeleton Creek, Enid, Oklahoma. EPA/600/30-85-044. Duluth Environmental Research Laboratory, Minnesota.

Norton, S.B. 1999. *Using biological monitoring data to distinguish among types of stress in streams of the Eastern Cornbelt Plains Ecoregion.* Ph.D. Dissertation, Georgetown University, Fairfax, VA.

Norton S.B., S.M. Cormier, M. Smith, and R.C. Jones. 2000. Can biological assessment discriminate among types of stress? A case study for the eastern cornbelt plains ecoregion. *Environ. Toxicol. Chem.* 19(4):1113-1119.

Ohio Environmental Protection Agency (OEPA). 1988a. *Biological criteria for the protection of aquatic life: Vol. II. users manual for biological assessment of Ohio surface waters.* Division of Water Quality Planning and Assessment, Ecological Assessment Section, Columbus, OH.

__________. 1988b. *Biological and water quality study of the Little Scioto River watershed*, Marion County, OH. OEPA Technical Report prepared by State of Ohio Environmental Protection Agency, Division of Surface Water, Columbus, OH.

__________. 1989a. *Addendum to: Biological criteria for the protection of aquatic life: Volume II. users manual for biological assessment of Ohio surface waters*. Division of Water Quality Planning and Assessment, Ecological Assessment Section, Columbus, OH.

__________. 1989b. *Biological criteria for the protection of aquatic life: Volume III. standardized field and laboratory methods for assessing fish and macroinvertebrate communities.* Division of Water Quality Planning and Assessment, Ecological Assessment Section, Columbus, OH.

__________. 1989c. *Manual of Ohio EPA surveillance methods and quality assurance practices.* Division of Environmental Services, Columbus, OH.

__________. 1992a. *Bottom sediment evaluation, Little Scioto River, Marion, Ohio.* Division of Water Quality Planning and Ecological Assessment Section, Columbus, OH.

__________. 1992b. *Biological and water quality study of the Ottawa River, Hog Creek, Little Hog Creek, and Pike Run.* OEPA Technical Report EAS/1992-9-7. Prepared by State of Ohio Environmental Protection Agency, Division of Surface Water, Columbus, OH.

__________. 1994. *Biological, sediment, and water quality study of the Little Scioto River, Marion, Ohio.* OEPA Technical Report EAS/1994-1-1. Division of Surface Water, Ecological Assessment Section, Columbus, OH.

Platt, J.R. 1964. Strong inference. *Science* 146:347-353.

Popper, K.R. 1968. *The logic of scientific discovery*. Harper and Row, New York.

Rankin, E.T. 1989. *The qualitative habitat evaluation index (QHEI): rationale, methods, and application*. State of Ohio Environmental Protection Agency, Division of Water Quality Planning and Assessment, Ecological Assessment Section, Columbus, OH.

__________. 1995. Habitat indices in water resource quality assessments. Pages 181-208 *in* W.S. Davis and T.P. Simon (editors). *Biological Assessment and Criteria.* Lewis Publishers, Boca Raton, Florida.

Rankin E., R. Miltner, C. Yoder and D. Mishne. 1999. *Association between nutrients, habitat, and the aquatic biota in Ohio rivers and streams.* Ohio EPA Technical bulletin MAS/1999-1. Ohio EPA, Columbus, OH.

Rosgen, D. 1996. *Applied river morphology.* Wildland Hydrology Books, Pagosa Springs, CO.

Roubal, W.T., T.K. Lallier, and D.C. Malins. 1977. Accumulation and metabolism of C-14 labeled benzene, naphthalene, and anthracene by young coho salmon (*Oncorhynchus kisutch*) and starry flounder (*Platichthys stellatus). Arch. Environ. Contam. Toxicol.* 5: 513-529.

Russo, R.C. 1985. Ammonia, nitrate and nitrite. Pages 455-471 *in* G.M. Rand and S.A. Petrocelli (editors). *Fundamentals of Aquatic Toxicology.* McGraw Hill, Washington, D.C.

Sheilds, F.D. Jr., S.S. Knight, and C.M. Cooper. 1998. Rehabilitation of aquatic habitats in warmwater streams damaged by channel incision in Mississippi. *Hydrobiologica* 382:63-86

Smith, V.H., G.D. Tilman, and J.C. Nekola. 1999. Eutrophication: impacts of excess nutrient inputs on freshwater, marine and terrestrial ecosystems. *Environmental Pollution* 100:179-196.

Susser, M. 1986a. Rules of inference in epidemiology. *Regulatory Toxicology and Pharmacology* 6:116-186.

________. 1986b. The logic of Sir Karl Popper and the practice of epidemiology. *Am. J. Epidemiol.* 124:711-718.

________. 1988. Falsification, verification and causal inference in epidemiology: Reconsideration in light of Sir Karl Popper's philosophy. Pages 33-58 *in* K.J. Rothman (ed.). *Causal Inference.* Epidemiology Resources Inc., Chestnut Hill, MA.

Suter, G.W., II. 1990. Use of biomarkers in ecological risk assessment. Pages 419-426 *in* J.F. McCarthy and L. L. Shugart (eds.). *Biomarkers of Environmental Contamination.* Lewis Publishers, Ann Arbor, Michigan.

__________. 1993. *Ecological risk assessment.* Lewis Publishers, Boca Raton, FL.

__________. 1998. Retrospective assessment, ecoepidemiology, and ecological monitoring. Pages 177-217 *in* P. Calow (ed.). *Handbook of Environmental Risk Assessment and Management.* Blackwell Scientific, Oxford, UK.

__________. 1999. Developing conceptual models for complex ecological risk assessments. *Human & Ecolog. Risk Assess.* 5:375-396.

Suter, G.W. II, J.W. Gillett, and S. Norton. 1994. Characterization of exposure. Chapter 4 *in Ecological Risk Assessment Issue Papers*. EPA/630/R-94/009. U.S. Environmental Protection Agency, Washington, D.C.

Tarplee, W.H. Jr., D.E. Louder, and A.J. Weber. 1971. *Evaluation of the effects of channelization on fish populations in North Carolina's coastal plain streams.* North Carolina Wildlife Resources Commission, Raleigh, NC.

Thornton, K.W., G.E. Saul, and D.E. Hyatt. 1994. *Environmental monitoring and assessment program assessment framework.* EPA/620/R-94/016. U.S. Environmental Protection Agency, Research Triangle Park, NC.

U.S. Environmental Protection Agency (USEPA). 1988a. *Generalized methodology for conducting industrial toxicity reduction evaluations.* EPA/600-2-88/070.

__________. 1988b. *Toxicity reduction evaluation protocol for municipal wastewater treatment plants.* EPA/600/2-88/062.

__________. 1988c. *Methods for aquatic toxicity identification evaluations: phase 1, toxicity characterization procedures* (EPA/600/3-88/034); *Phase 2, toxicity identification procedures* (EPA/600/3-88/035); and *Phase 3, toxicity confirmation procedures*. EPA/600/3-88/036.

__________. 1991a. *The watershed protection approach: An overview.* Office of Water. Washington, DC. EPA 503/9-92-001.

__________. 1991b. *Methods for aquatic toxicity identification evaluations, phase I toxicity characterization procedures, 2nd ed.* U.S. Environmental Protection Agency, Office of Research and Development, Environmental Research Laboratory, Duluth, MN. EPA/600/6-91/003.

__________. 1993a. *Methods for aquatic toxicity identification evaluations, phase II toxicity identification procedures for samples exhibiting acute and chronic toxicity.* EPA/600/R-92/080. (NTIS: PB94-114907).

__________. 1993b. *Methods for aquatic toxicity identification evaluations, phase III toxicity confirmation procedures for samples exhibiting acute and chronic toxicity*. U.S. Environmental Protection Agency, Office of Research and Development, Environmental Research Laboratory, Duluth, MN. EPA/600/R-92/081.

__________. 1994. *Interim guidance on determination and use of water-effect ratios for metals.* U.S. Environmental Protection Agency, Office of Science and Technology, Washington, DC. EPA/823/B-94/001.

__________. 1996a. *Summary report of workshop on Monte Carlo analysis.* Office of Research and Development, Risk Assessment Forum, Washington, DC. EPA/630/R-96/010.

__________. 1996b. Calculation and evaluation of sediment effect concentrations for the amphipod *Hyallela azteca* and the midge *Chironomus riparius*. Assessment and Remediation of Contaminated Sediments (ARCS) Program. Great Lakes National Program Office, Chicago, IL. EPA 905-R96-008.

__________. 1997. *Guidelines for preparation of the comprehensive state water quality assessments (305(b) Reports) and electronic updates: Report contents.* U.S. Environmental Protection Agency, Office of Water. Washington, DC 20460. EPA-841-B-97-002A.

__________. 1998a. *Guidelines for ecological risk assessment.* Office of Research and Development. Risk Assessment Forum, Washington, D.C. EPA/630/R-95/002F.

__________. 1998b. *1998 update of ambient water quality criteria for ammonia.* Office of Water, Washington, DC. EPA 822-R-98-008.

__________. 1999. *Report of the workshop on selecting input distributions for probabilistic assessments.* Office of Research and Development, Risk Assessment Forum, Washington, DC. EPA/630/R-98/004

Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell, and C.E. Cushing. 1980. The river continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences* 37:130-137.

Varanasi, U., J.E. Stein, M. Nishimoto, and T. Hom. 1983. Benzo[a]pyrene metabolites in liver, muscle, gonads, and bile of adult English sole (*Parophrys vetulus*). Pages 1221-1234 *in* Cooke, M. and A.J. Dennis, eds. *Polynuclear Aromatic Hydrocarbons: Formation, Metabolism, and Measurement*. Battelle, Columbus, OH, USA.

Woodman, J.N., and E.B. Cowling. 1987. Airborne chemicals and forest health. *Environ. Sci.* 21:120-126.

Yerushalmy, J., and C.E. Palmer. 1959. On the methodology of investigations of etiologic factors in chronic disease. *J. Chronic Disease* 10(1):27-40.

Yoder, C.O., and E.T. Rankin. 1995a. Biological criteria program development and implementation in Ohio. Pages 109-144 *in* W.S. Davis and T.P. Simon, eds. *Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making.* Lewis Publishers, Boca Raton, FL.

__________. 1995b. Biological response signatures and the area of degradation value: New tools for interpreting multi-metric data. Pages 236-286 *in Biological Assessment and Criteria: Tools for Water Resource Planning and Decisionmaking* Lewis Publishers, Boca Raton, FL..

Yount, J.D., and G.J. Niemi. 1990. Recovery of lotic communities and ecosystems from disturbance; A narrative review of case studies. *Environmental Management* 14:547-569.

Index

N

O

P

Q

R

S

Use of Biological Information to Better Define Designated Aquatic Life Uses in State and Tribal Water Quality Standards: Tiered Aquatic Life Uses

DISCLAIMER

The discussion in this draft document is intended solely to provide information on advancements in the field of bioassessments and on current State and Tribal practices using bioassessments to define their designated aquatic life uses. The statutory provisions and U.S. EPA regulations described in this document contain legally binding requirements. This document is not a regulation itself, nor does not it change or substitute for those provisions and regulations. Thus, it does not impose legally binding requirements on U.S. EPA, States, or the regulated community. This document does not confer legal rights or impose legal obligations upon any member of the public.

While U.S. EPA has made every effort to ensure the accuracy of the discussion in this document, the obligations of the regulated community are determined by statutes, regulations, or other legally binding requirements. In the event of a conflict between the discussion in this document and any statute or regulation, this document would not be controlling.

The general description provided here may not apply to a particular situation based upon the circumstances. Interested parties are free to raise questions and objections about the substance of this document and the appropriateness of the application of the information presented to a particular situation. U.S. EPA and other decision-makers retain the discretion to adopt approaches on a case-by-case basis that differ from those described in this document where appropriate.

Mention of trade names or commercial products does not constitute endorsement or recommendation for their use.

This is a living document and may be revised periodically. U.S. EPA welcomes public input on this document at any time.

Preface

Our Nation's waters are a valuable ecological resource. Protecting them begins with State and authorized Tribal adoption of water quality standards. This draft document, the *Use of Biological Information to Better Define Designated Aquatic Life Uses in State and Tribal Water Quality Standards: Tiered Aquatic Life Uses*, provides up-to-date information on practical, defensible approaches to help States and Tribes more precisely define designated aquatic life uses in their water quality standards. Biologically-based tiered aquatic life uses, based on the scientific model presented in this document, can help States and Tribes develop aquatic life uses that more precisely describe the existing and potential uses of a waterbody and then use bioassessments to help measure attainment of the uses.

Biologically-based tiered aquatic life uses coupled with numeric biological criteria provide a direct measure of the aquatic resource that is being protected. The condition of the biota reflects the cumulative response of the aquatic community to individual or multiple sources of stress – an environmental outcome measure. The technical approaches described in this document support U.S. EPA's Environmental Indicators Initiative to move the Agency closer to a performance-based rather than process-based environmental protection system (http://www.epa.gov/indicators). Launched in November 2001, the Environmental Indicators Initiative responds to the President's call to have agencies and departments manage for results by measuring environmental outcomes.

This document is a compilation of the tools, practices, and experiences of State and Tribal scientists who have used biological information to more precisely define their aquatic life uses. The presented model brings biological condition and stressor information together to inform decisions on use designation. The document fulfills a commitment in the U.S. EPA Water Quality Standards Strategy to provide technical support, outreach, training, and workshops to assist States and Tribes with designated uses, including use attainability analyses and tiered aquatic life uses (EPA-823-R-03-010, Strategic Action #7, Milestone #2).

U.S. EPA encourages States and Tribes to incorporate biological information into their decisions. U.S. EPA believes the use of bioassessments will help improve water quality protection. The information in this document can help States and Tribes use bioassessments to more precisely define their aquatic life uses and communicate this information to the public. U.S. EPA is making this document available so States and Tribes can pilot a bioassessment-based tiered approach to defining their designated aquatic life uses. If you choose to undertake a pilot, U.S. EPA would appreciate hearing about your experience. We are interested in feedback on the following questions:

- Is this document helpful in addressing current issues in your program?
- Does this document address the technical challenges in your Region, State, or Tribe?
- How can this document be improved to help you develop tiered aquatic life uses in your program?
- What additional information would be helpful to you?

Should you have any questions or wish to provide feedback, please contact Susan K. Jackson via email at Jackson.Susank@epa.gov or at the following address:

Tiered Aquatic Life Uses Document
Attn: Susan K. Jackson
Health and Ecological Criteria Division (4304T)
Office of Science and Technology
U.S. EPA, Office of Water
1200 Pennsylvania Avenue
Washington, DC 20460

Executive Summary

This document provides up-to-date information on how States and Tribes can use biological information to more precisely define designated aquatic life uses for their waters. Thirty years ago, under the Clean Water Act (CWA), States and Tribes were required to adopt in their water quality standards, where attainable, designated uses that included the protection and propagation of fish, shellfish, and wildlife. During the 1970s, the biological goals adopted into State or Tribal water quality standards as designated aquatic life uses may have been appropriately general (e.g., "aquatic life as naturally occurs") given the limited data available and the state of the science. However, while such general use classifications meet the requirements of the Clean Water Act and the implementing federal regulations, they may constitute the beginning, rather than the end, of appropriate use designations. Improved precision may result in more efficient and effective evaluation of attainment of condition and utilization of restoration resources. Finally, improved precision in uses can enhance demonstrating progress towards management goals. In the years since the CWA was passed, considerable advancements have been made in the science of aquatic ecology and in biological monitoring and assessment methods. This document summarizes these advancements and provides a scientific model that States and Tribes can use to refine their designated uses in a manner that can improve their water quality assessment and management.

This document was developed based on the technical expertise and practical experience of State and Tribal scientists. In 2000, the U.S. EPA convened a technical expert workgroup, including State and Tribal scientists, to identify scientifically sound and practical approaches to help States and Tribes provide more specificity in their designated aquatic life uses. The workgroup developed a scientific model, the Biological Condition Gradient (BCG), which describes biological response to increasing levels of stressors. The model describes how ten attributes of aquatic ecosystems change in response to increasing levels of stressors. The attributes include several aspects of community structure, organism condition, ecosystem function, and spatial and temporal attributes of stream size and connectivity. The gradient can be considered analogous to a field-based dose-response curve where dose (x-axis) = increasing levels of stressors and response (y-axis) = biological condition. The BCG differs from the standard dose-response curve, in that the BCG does not represent the laboratory response of a single species to a specified dose of a known chemical, but rather the *in situ* response of the biota to the sum of stresses it is exposed to. The BCG is divided into six tiers of biological condition along the stressor-response curve, ranging from observable biological conditions found at no or low levels of stress to those found at high levels of stressors. The model provides a common framework for interpreting biological information regardless of methodology or geography. When calibrated to a regional or state scale, States and Tribes can use this model to more precisely evaluate the current and potential biological condition of their waters and use that information to inform their decisions on aquatic life designations. Additionally, States and Tribes can use this interpretative model to more clearly and consistently communicate these decisions to the public.

Maine and Ohio have adopted biologically-based tiered aquatic life uses in their WQS and have over twenty years experience implementing this type of use designation approach. Both Maine and Ohio developed and adopted tiered aquatic life uses for similar reasons: 1) to incorporate ecologically relevant endpoints into decisions; 2) to inform water quality management decisions; 3) to quantify water quality improvements; and 4) to merge the design and practice of monitoring and assessment with the development and implementation of their water quality standards. Maine and Ohio scientists have identified a sequence of steps and milestones that U.S. EPA has compiled as a template that other States and Tribes may use to develop biologically-based tiered uses. Examples from Maine and Ohio are included in this document to illustrate how they used biological data to establish tiered uses and the programmatic gains from having done so.

The U.S. EPA encourages States and Tribes to incorporate biological information into their decisions. The U.S. EPA believes that the use of biological information can help improve water quality protection. Currently, States and Tribes that use biological data as part of their assessment program apply some type of tiered aquatic life use to guide their interpretation of their biological data. States and Tribes have either explicitly adopted tiers directly into their water quality standards as designated uses, or used tiers in monitoring and assessment of their surface waters. This document provides examples of practical and scientifically sound approaches to using biological information to tier designated aquatic life uses.

FOREWORD
Why is U.S. EPA publishing this document?

In the more than 30 years since the Clean Water Act (CWA) was passed, there has been considerable progress in the science of aquatic ecology and in the development of biological monitoring and assessment techniques. During the 1970s, the biological goals adopted into State or Tribal water quality standards as designated aquatic life uses may have been appropriately general (e.g., "aquatic life as naturally occurs") given the limited data available and the state of the science. However, while such general use classifications meet the requirements of the Clean Water Act and the implementing federal regulations, they may constitute the beginning, rather than the end, of appropriate use designations. Improved precision may result in more efficient and effective evaluation of attainment of condition and utilization of restoration resources. Finally, improved precision in uses can enhance demonstrating progress towards management goals. Tiered aquatic life uses, based on the biological condition gradient model presented in this document, can help States and Tribes to better define and develop more precise, scientifically defensible aquatic life uses that account for the natural differences between waterbodies and should result in more appropriate levels of protection for specific waterbodies.

States and Tribes have created different use classification systems ranging from a straightforward replication of the general uses identified in the CWA (e.g., protection and propagation of fish, shellfish, and wildlife; recreation; agriculture; industrial and other purposes, including navigation) to more complex systems that express designated uses in more specific terms or establish classifications which identify different levels of protection. For example, some States designate general "aquatic life" uses, while others subcategorize waters based on the expected biological assemblage. Some have established tiers representing different levels of biological condition (e.g., excellent, good, fair). Although a variety of defensible approaches have evolved and become established in State and Tribal programs, current U.S. EPA regulations are not specific about the level of precision States or Tribes must achieve in designating uses. This document is designed to help inform States and Tribes how to better define and improve the precision of their designated uses.

Over the past thirty years, both the state of aquatic science and the application of the science in State and Tribal water programs have advanced. Major areas of uncertainty in water management, such as distinguishing between natural variability and effects of stressors on aquatic systems as well as determining the appropriate level of protection for individual waterbodies, are being addressed. Many States and Tribes now use biological information to directly assess the biological condition of their aquatic resources (U.S. EPA 2002a). Three States have formally adopted biologically-based tiered aquatic life uses in their water quality standards. "Lessons learned" from two of these States indicate that implementation of tiered aquatic life uses supports more appropriate levels of protection for individual waters by promoting uses and criteria that are neither over- nor under-protective. U.S. EPA now recognizes that the States having implemented tiered aquatic life uses have significantly benefited from the approach. The use designation process needs to clearly articulate and differentiate intended levels of protection with enough specificity so that 1) decision makers can appropriately develop and implement their water quality standards on a site, reach, or watershed specific basis and 2) the public can understand, identify with, and influence the goals set for waters.

In 2001, the National Research Council (NRC) published its report on *Assessing the TMDL Approach to Water Quality Management* (NRC 2001). In the report, the NRC recommended tiering designated uses as an essential step in setting water quality standards and improving decision-making. The NRC, finding that the Clean Water Act's goals (i.e., "fishable," "swimmable") are too broad to serve as operational statements of designated use, recommended greater specificity in defining such uses. For example, rather than stating that a waterbody needs to be "fishable," the designated use would ideally describe the expected fish assemblage or population (e.g., cold water fishery, warm water fishery, or salmon, trout, bass, etc.) as well as the other biological assemblages necessary to support that fish population.

Additionally, the NRC recommended that biological criteria should be used in conjunction with physical and chemical criteria to determine whether a waterbody is meeting its designated use. The NRC described a "position of the criterion" framework, which reflects how representative a criterion is of a designated use according to its position along a conceptual causal pathway (Figure F-1). This alignment is comparable to that of performance (indicators of point source quality) versus impact standards (indicators of resource condition) (Courtemanch et al. 1989), or of stressor and exposure (effluent, chemical, and physical parameters) in contrast to response indicators (biological) (Yoder and Rankin 1998). In Figure F-1, stressor indicators correspond to box 1 and were termed effluent standards by the NRC. Pollutant-specific indicators that function as indicators of exposure and stress correspond to box 2. Biological indicators show responses to stress and exposure and correspond to box 3. Because designated uses are written in qualitative, narrative terminology, the challenge is to relate a criterion to the designated use. Establishing this relationship is easier as the criterion is positioned closer to the designated use, thus the NRC recommendation on the use of biological information to help determine more appropriate aquatic life uses and to couple the narrative use statements with quantitative methods. The "position of criterion" concept provides a useful construct for considering the relationship of water quality criteria (biological, chemical, and physical) to the designated uses they are intended to protect.

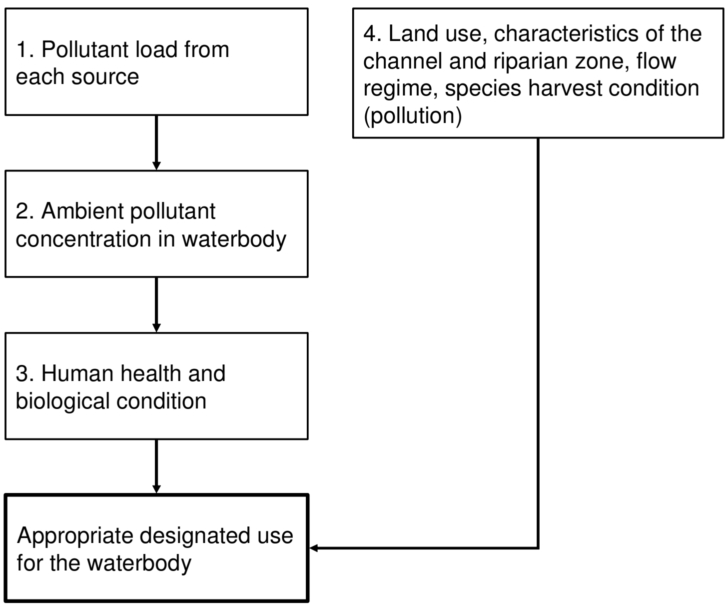


FIGURE F-1. Types of water quality criteria and their position relative to designated uses (after NRC 2001).

To help States and Tribes more precisely define use descriptions, there is a need to incorporate current scientific understanding of aquatic ecology and the appropriate use of monitoring data. To this end, the U.S. EPA convened a technical expert workgroup to identify scientifically sound and practical approaches that would help States and Tribes provide more specificity in their designated aquatic life uses. The workgroup met four times between 2000 and 2003. The workgroup, composed primarily of U.S. EPA, State, and Tribal scientists, also included research scientists from the U.S Geological Survey (USGS), the academic community, and the private sector. The workgroup was asked to base their recommendations on "lessons learned" from State and Tribal water programs in the development and the application of biologically-based aquatic life uses, bioassessments, and biocriteria. The workgroup developed a scientific model, the Biological Condition Gradient (BCG), which describes graduated tiers of biological response to increasing levels of stressors. This model was developed and tested through a series of data exercises using a diverse array of data sets. States and Tribes can use the BCG to more precisely define and set appropriate designated aquatic life uses for their waters.

During the final workgroup meeting in 2003, State and Tribal members discussed their current thinking on how using biological information to tier designated aquatic life uses could benefit their water quality management programs. The main reasons discussed included biologically-based tiered uses could help:

- set ecologically-based aquatic life goals for waterbodies;
- establish a consistent approach for identifying attainable, incremental restoration goals that are grounded in the concept of biological integrity;
- provide a framework that better relates traditional water quality criteria (stressor and exposure variables) and biological criteria (response variables) in determining use attainment, thus strengthening stressor/response models implicit in designated uses and criteria in water quality standards;
- better link monitoring and assessment with water quality standards; and
- prioritize management actions that result in the more effective use of resources.

When asked about the significant value-added outcomes of these benefits to their water programs, States and Tribes workgroup members anticipated being able to make more scientifically defensible listings of impaired waters as well as enhance identifying and protecting high quality waters. For several States, biologically-based tiered uses may help in the transition from reliance on current conditions in developing designated uses to being able to better consider the potential for improvement. Another important added value anticipated by all State and Tribal representatives was the ability to communicate more effectively with program managers, the public, and key stakeholders. Workgroup members expressed the opinion that biologically-based aquatic life uses could help maximize the return on their monitoring and assessment efforts by eliminating a major source of uncertainty in water quality management by 1) accounting for natural variability in aquatic systems and 2) helping to specify an appropriate level of protection for a waterbody that includes consideration of the system's potential for improvement.

Biologically-based aquatic life uses, as described in this document, are a natural evolution that reflects an improved understanding of surface waters resulting from more than 20 years of assessment data. The proposed approach will help better integrate the science of aquatic ecology into Water Quality Standards. This document represents the culmination of four years of workgroup deliberations, including four workgroup meetings and two workshops to "road test" the BCG model. Based on the collective experience of the workgroup members, the science and methods in the fields of biological assessments and criteria have progressed sufficiently over the past thirty-five years to support the use of biological information to tier designated aquatic life uses in State and Tribal water quality standards.

Acknowledgments

U.S. EPA PROJECT LEAD
Susan Jackson, U.S. EPA Office of Science and Technology

WRITING & EDITING TEAM
David Allan, University of Michigan; Margo Andrews, Tetra Tech, Inc.; Michael Barbour, Tetra Tech, Inc.; Jan Cibrowski, University of Windsor; Maggie Craig, Tetra Tech, Inc.; Susan Davies, Maine Department of Environmental Protection; Tom Gardner, U.S. EPA; Jeroen Gerritsen, Tetra Tech, Inc.; Charles Hawkins, Utah State University; Robert Hughes, Oregon State University; Susan Jackson, U.S. EPA; Lucinda Johnson, Natural Resources Research Institute, University of Minnesota – Duluth; Phil Larsen, U.S. EPA; JoAnna Lessard, Tetra Tech, Inc.; Abby Markowitz, Tetra Tech, Inc.; Dennis McIntyre, Great Lakes Environmental Center; Jerry Niemi, Natural Resources Research Institute, University of Minnesota – Duluth; Dave Pfeifer, U.S. EPA; Ed Rankin, Center for Applied Bioassessment and Biocriteria; Tom Wilton, Iowa Department of Natural Resources; Chris Yoder, Midwest Biodiversity Institute

TIERED AQUATIC LIFE USES WORKGROUP

U.S. EPA Chair: Susan Jackson, U.S. EPA Office of Science and Technology
State Chair: Susan Davies, Maine Department of Environmental Protection

STATE AND TRIBAL WORKGROUP MEMBERS

Arizona Department of Environmental Quality – Patti Spindler

California Department of Fish and Game – Jim Harrington

Colorado Department of Public Health and Environment – Robert McConnell, Paul Welsh

Florida Department of Environmental Protection – Leska Fore, Russ Frydenborg, Ellen McCarron, Nancy Ross

Idaho Department of Environmental Quality – Mike Edmondson, Cyndi Grafe*

Kansas Department of Health and Environment – Bob Angelo, Steve Haslouer, Brett Holman

Kentucky Department for Environmental Protection – Greg Pond*, Tom VanArsdall

Maine Department of Environmental Protection – David Courtemanch, Susan Davies

Maryland Department of the Environment – Joseph Beaman, Richard Eskin, George Harmon

Minnesota Pollution Control Agency – Greg Gross

Mississippi Department of Environmental Quality – Leslie Barkley, Natalie Guedon

Montana Department of Environmental Quality – Randy Apfelbeck, Rosie Sada

Nevada Division of Environmental Protection – Karen Vargas

North Carolina Department of Environment and Natural Resources – David Lenat, Trish MacPherson

Ohio Environmental Protection Agency – Jeff DeShon, Dan Dudley

Ohio River Valley Water Sanitation Commission – Erich Emery

Oregon Department of Environmental Quality – Doug Drake, Rick Hafele

Pyramid Lake Paiute Tribe – Dan Mosley

Texas Commission on Environmental Quality – Charles Bayer

Vermont Department of Environmental Conservation – Doug Burnham, Steve Fiske

Virginia Department of Environmental Quality – Alexander Barron, Larry Willis

Washington State Department of Ecology – Robert Plotnikoff

Wisconsin Department of Natural Resources – Joe Ball, Ed Emmons, Robert Masnado, Greg Searle, Michael Talbot, Lizhu Wang**

U.S. EPA
Office of Water: Chris Faulkner, Thomas Gardner, Susan Holdsworth, Susan Jackson, Kellie Kubena, Douglas Norton, Christine Ruff, Robert Shippen, Treda Smith, William Swietlik

Regional Offices:
Region 1: Peter Nolan
Region 2: Jim Kurtenbach
Region 3: Maggie Passmore
Region 4: Ed Decker, Jim Harrison, Eve Zimmerman
Region 5: Ed Hammer, David Pfeifer
Region 6: Philip Crocker, Charlie Howell
Region 7: Gary Welker
Region 8: Tina Laidlaw, Jill Minter
Region 9: Gary Wolinsky
Region 10: Gretchen Hayslip

Office of Environmental Information: Wayne Davis

Office of Research and Development: Karen Blocksom, Susan Cormier, Phil Larsen, Frank McCormick, Susan Norton, Danielle Tillman, Lester Yuan

USGS
Evan Hornig*, Ken Lubinski

SCIENTIFIC COMMUNITY
David Allan, University of Michigan
Michael Barbour, Tetra Tech, Inc.
David Braun, The Nature Conservancy
Jeroen Gerritsen, Tetra Tech, Inc.
Richard Hauer, University of Montana
Charles Hawkins, Utah State University
Robert Hughes, Oregon State University
James Karr, University of Washington
Dennis McIntyre, Great Lakes Environmental Center
Ed Rankin, Center for Applied Bioassessment and Biocriteria
Jan Stevenson, Michigan State University
Denice Wardrop, Pennsylvania State University
Chris Yoder, Midwest Biodiversity Institute

BCG STEERING COMMITTEE
Michael Barbour, Susan Davies, Robert Hughes, Susan Jackson, Phil Larsen, Dennis McIntyre, Susan Norton, Maggie Passmore, Jan Stevenson, Chris Yoder, Lester Yuan

STRESSOR GRADIENT STEERING COMMITTEE
David Allan, Michael Barbour, Jan Cibirowski, Jim Harrison, Robert Hughes, Lucinda Johnson, JoAnna Lessard, Jerry Niemi, Doug Norton, Ed Rankin, Tom Wilton

*Now with U.S. EPA
**Now with Michigan Department of Natural Resources

Use of Biological Information to Better Define Designated Aquatic Life Uses in State and Tribal Water Quality Standards: Tiered Aquatic Life Uses

TABLE OF CONTENTS

Tables, Figures, and Case Examples

Tables

Figures

Case Examples

Introduction

This chapter provides the background and rationale for using biological information to tier designated aquatic life uses and better define them in State and Tribal water quality standards. Ideally, the use designation process clearly articulates and differentiates intended levels of protection with enough specificity so that 1) decision makers can appropriately develop and implement their water quality standards on a reach or watershed specific basis; and 2) the public can understand, identify with, and influence the goals set for waters. In 2000, the U.S. EPA convened a technical expert workgroup, including State and Tribal scientists, to identify existing scientifically sound and practical approaches using biological information to better define aquatic life uses. The workgroup produced a scientific model, the Biological Condition Gradient (BCG), for interpreting biological response to increasing levels of stressors. The workgroup's findings are consistent with The National Research Council's call for greater specificity in water quality standards that can result in improved decision-making (NRC 2001). The BCG is intended to help States and Tribes develop more precise aquatic life uses that should result in more appropriate levels of protection for their surface waters.

CHAPTER 1. WHAT ARE TIERED AQUATIC LIFE USES?

Designated aquatic life uses are State or Tribal descriptions of the biological goals for their waterbodies. Tiered aquatic life uses (TALUs) use biological information to more precisely define these goals relative to natural conditions. Bioassessments can then be used to measure attainment of the goals. U.S. EPA's current thinking is that a system of tiered uses could:

- accommodate observable differences in expected biological condition in waterbodies in different ecological regions;
- provide an objective means of describing the biological potential for a specific waterbody;
- recognize and accommodate observable differences in biological potential among waters with different types and levels of stressors;
- reflect an understanding of the relationship between stressors and biological community response;
- guide selection of environmental indicators for monitoring and assessment and make full use of available biological data; and
- articulate a stressor-response model that maximizes the likelihood of success of water quality management actions based on water quality standards (assessment, 303(d) listings/TMDLS, NPDES permits).

Tiered aquatic life uses are based on general observations about aquatic communities that have become central to aquatic ecology and consistent with 30 years of empirical observations. These are:

- surface waters and the biological communities they support are predictably and consistently different in different parts of the country *(classification along a natural gradient, ecological region concept)*;
- within the same ecological regions, different types of waterbodies (e.g., headwaters, streams, rivers, wetlands) support predictably and consistently different biological communities *(waterbody classification)*;
- within a given class of waterbodies, observed biological condition in a specific waterbody is a function of the level of stress (natural and anthropogenic) that the waterbody has experienced *(the biological condition gradient discussed in this document)*;

- similar stressors at similar intensities produce predictable and consistent biological responses in waters within a class, and those responses can be detected and quantified in terms of deviation from an expected condition *(reference condition)*; and
- waterbodies exposed to higher levels of stressors will have lower biological performance compared to the reference condition than those waters experiencing lower levels of stress (*the biological condition and stressor gradients discussed in this document).*

The first three sections of this chapter provide the statutory and regulatory background of water quality standards, emphasizing the role of designated aquatic life uses. Section 1.4 explores how tiered biologically-based definitions can help set more appropriate and precise designated aquatic life uses in State and Tribal water quality standards. The next two sections discuss the primary products of the technical workgroup charged with identifying existing scientifically sound and practical approaches to help States and Tribes to better define and provide more precision in their designated aquatic life uses. Chapter 1 concludes with a summary of key points, organization of the document, and related technical support documents.

1.1 The CWA goals and objectives for aquatic life

One objective of the 1972 Clean Water Act (CWA) is to restore and maintain the chemical, physical, and biological integrity of the Nation's waters (CWA sec 101a). In the scientific literature, an aquatic system with chemical, physical, and biological integrity has been described as being capable of "supporting and maintaining a balanced, integrated, adaptive community of organisms having a composition and diversity comparable to that of the natural habitats of the region" (Frey 1977). Over the intervening years, our understanding of how to define and measure the integrity of aquatic systems has advanced. The term *integrity* has been further refined in the literature to mean a balanced, integrated, adaptive system having a full range of ecosystem elements (genes, species, assemblages) and processes (mutation, demographics, biotic interactions, nutrient and energy dynamics, metapopulation dynamics) expected in areas with no or minimal human influence (Karr 2000). The aquatic biota residing in a waterbody are the result of complex and interrelated chemical, physical, and biological processes that act over time and on multiple scales (e.g., instream, riparian, landscape) (Karr et al. l986, Yoder l995). By directly measuring the condition of the aquatic biota, we are able to more accurately define the aquatic community that is the outcome of all these factors.

To help achieve the integrity objective, the CWA also established an interim goal for the protection and propagation of fish, shellfish, and wildlife and recreation in and on the water. The protection and propagation interim goal for aquatic life has been interpreted by U.S. EPA to include the protection of the full complement of aquatic organisms residing in or migrating through a waterbody. As explained in U.S. EPA's *Questions and Answers on Antidegradation*, the protection afforded by water quality standards includes the representative aquatic community (e.g., fish, benthic macroinvertebrates, and periphyton):

> "The fact that sport or commercial fish are not present does not mean that the water may not be supporting an aquatic life protection function. An existing aquatic community composed entirely of invertebrates and plants, such as may be found in a pristine tributary alpine stream, should be protected whether or not such a stream supports a fishery. Even though the shorthand expression 'fishable/swimmable' is often used, the actual objective of the Act is to restore the chemical, physical and biological integrity of our Nation's waters (Section 101(a)). The term 'aquatic life' would more accurately reflect the protection of the aquatic community that was intended in Section 101(a)(2) of the Act." (Appendix G, EPA-823-B-94-005)

The representative community of aquatic organisms residing in, or migrating through, a waterbody will vary depending on the waterbody type. For example, fish, benthic macroinvertebrates, and, increasingly, periphyton are aquatic assemblages typically measured by States and Tribes when assessing streams and rivers. In headwater streams and many wetlands, amphibians are an important component of the biotic community and fish may be absent.

1.2 WQS statutory and regulatory background

Section 101(a) of the CWA establishes broad national goals and objectives such as the chemical, physical, and biological integrity objective. Other sections of the CWA establish the programs and authorities for implementation of those goals and objectives. Section 303(c) sets up the basis of the current water quality standards program. Water quality *standards* (WQS) are parts of State (or, in certain instances, federal) law that define the water quality goals of a waterbody, or parts of a waterbody, by designating the use or uses of the waterbody and by setting criteria necessary to protect the uses. The standards also include an antidegradation policy consistent with 40 CFR Part 131.12.

Although the CWA gives the U.S. EPA an important role in determining appropriate minimum levels of protection and providing national oversight, it also gives considerable flexibility and discretion to States and Tribes to design their own programs and establish levels of protection beyond the national minimums. Section 303 directs States and authorized Tribes to adopt water quality standards to protect public health or welfare, enhance the quality of water, and serve the purposes of the Clean Water Act. "Serve the purposes of the Act" (as defined in Sections 101(a), 101(a)(2), and 303(c) of the CWA) means that water quality standards should 1) include provisions for restoring and maintaining chemical, physical, and biological integrity of State and Tribal waters, 2) provide, wherever attainable, water quality for the protection and propagation of fish, shellfish, and wildlife and recreation in and on the water (i.e., "fishable/swimmable"), and 3) consider the use and value of State and Tribal waters for public water supplies, propagation of fish and wildlife, recreation, agricultural and industrial purposes, and navigation. Further requirements for water quality standards are at 40 CFR Part 131.

State WQS provide the foundation for water quality-based pollution control programs. With the public participating in their adoption (see 40 CFR 131.20), such standards serve the dual purposes of establishing the water quality goals for a specific waterbody, and serving as the regulatory basis for the establishment of water quality-based treatment controls and strategies beyond the technology-based levels of treatment required by Sections 301(b) and 306 of the CWA.

A waterbody's *designated use(s)* are those uses specified in water quality standards, whether or not they are being attained (40 CFR 131.3(f)). The "use" of a waterbody is the most fundamental description of its role in the aquatic and human environments. All of the water quality protections established by the CWA follow from the waterbody's designated use. As designated uses are critical in determining the water quality criteria that apply to a given waterbody, determining the appropriate designated use is of paramount importance in establishing criteria that are appropriately protective of that designated use.

Section 131.10 of the regulation describes States' and authorized Tribes' responsibilities for designating and protecting uses. The regulation:

- requires that States and Tribes specify the water uses to be achieved and protected,
- requires protection of downstream uses,
- allows for sub-category and seasonal uses,
- sets out minimum attainability criteria,
- lists six factors of, which at least one must be satisfied to justify removal of designated uses that are not existing uses,
- prohibits removal of existing uses,
- requires upgrading of uses that are presently being attained but not designated, and
- establishes conditions and requirements for conducting use attainability analyses.

In addition, the regulations effectively establish a "rebuttable presumption" that the uses of protection and propagation of fish, shellfish, and wildlife and recreation in and on the water are attainable and should apply to a waterbody, unless it has been affirmatively demonstrated that such uses are not attainable.

40 CFR 131.10(a) requires that States specify appropriate water uses to be achieved and protected. The classification of the waters of the State must take into consideration the use and value of water for public water supplies, protection and propagation of fish, shellfish, and wildlife, recreation in and on the water, and agricultural, industrial, and other purposes, including navigation. Changing designated uses for a specific waterbody requires a change in the water quality standards. Like all new and revised State and Tribal water quality standards, these changes are subject to U.S. EPA review and approval (see 40 CFR 131.21).

Where appropriate, a State may subcategorize or refine the aquatic life use designations for the receiving water. States may adopt subcategories of a use and set the appropriate criteria to reflect varying needs of such subcategories of uses, for instance, to differentiate between coldwater and warmwater fisheries (see 40 CFR 131.10(c)). States may also adopt seasonal uses (40 CFR 131.10(f)). If seasonal uses are adopted, water quality criteria should reflect the seasonal uses; however, such criteria shall not preclude the attainment and maintenance of a more protective use in another season.

Water quality *criteria* are elements of State WQS expressed as constituent concentrations, levels, or narrative statements representing a quality of water that supports a particular use. When criteria are met, water quality will generally protect the designated use (40 CFR 131.3). While some States have adopted a variety of criteria expressed as constituent concentration levels (or *numeric* criteria) for various pollutants for the protection of aquatic life, all States have adopted criteria expressed as narrative statements (or *narrative* criteria). Once adopted into standards, criteria can serve as the basis for 1) regulatory controls on point sources, 2) measuring attainment of standards and the effectiveness or programs, and 3) watershed planning.

Section 304(a) criteria are developed by the U.S. EPA under authority of section 304(a) of the CWA based on the latest scientific information on the relationship that a constituent concentration, level, or measure has on a particular aquatic species and/or human health. This information is issued periodically to the States as guidance for use in developing criteria. In adopting criteria to protect their designated uses, States may establish criteria based on 1) section 304(a) guidance, 2) section 304(a) guidance modified to reflect site-specific conditions, or 3) other scientifically defensible methods.

1.3 The role of designated aquatic life uses in Water Quality Standards

It is in designating uses that States and Tribes establish the environmental goals for their water resources and then measure attainment of these goals. In designating uses, a State or Tribe weighs the environmental, social, and economic consequences of its decisions. The regulation allows the State or Tribe, with public participation, some flexibility in weighing these considerations and adjusting these goals over time. However, reaching a conclusion on the uses that appropriately reflect the current and potential future uses for a waterbody, determining the attainability of those goals, and appropriately evaluating the consequences of a designation can be a difficult and controversial task.

A principal function of designated uses in water quality standards is to communicate the desired state of surface waters to water quality managers, the regulated community, and the interested public. An effective designated use system is one that translates readily into indicators (e.g., numeric water quality criteria, biological indexes) that respond in predictable ways to stress and can be evaluated using data collected from the waterbody. Experience with implementation of various State designated use systems suggests that, regardless of the system selected, States that use biological data as part of their assessment program apply some type of refined, or tiered, aquatic life use approach to guide interpretation of their biological data. States have either made this *explicit* by adopting the tiers directly into their water quality standards as designated uses or *implicit* by using tiers in their monitoring and assessment protocols.

Although the benefits of more specificity may apply to any of the designated uses described in CWA section 303, it may be most relevant for aquatic life uses. Aquatic communities can vary significantly from waterbody to waterbody. One major challenge in assigning designated uses for aquatic life to surface waters is separating the natural variability that is a function of stream type (e.g., naturally coldwater vs. warmwater stream) and location (ecoregion) from the variability that results from exposure to stressors. By accounting for natural variability in aquatic systems, biologically-based tiered aquatic life uses eliminate a major source of uncertainty and error in water quality management efforts.

1.4 State and Tribal experiences with tiered aquatic life uses

Over the years, States and Tribes have created many different use classification systems ranging from a straightforward replication of the uses specifically listed in section 303 of the CWA, to more complex systems that express designated uses in very specific terms or that establish subclassifications identifying different levels of protection. Some States designate general "aquatic life" uses while others list a variety of subcategories based on a range of aquatic community types, including descriptions of core aquatic species representative of each subcategory (e.g., coldwater and warmwater fisheries). Many States also have narrative biological criteria, which is often a general statement such as "aquatic life communities shall be maintained similar to aquatic life as naturally occurs." Single thresholds for attainment of these general uses and narrative biological criteria are established with numeric biological criteria. For example, many State water quality agencies interpret narrative general use statements using an index (e.g., Index of Biotic Integrity (IBI)) (Karr et al. 1987, Karr 1990, Gibson et al. 1996, U.S. EPA 2002a). The index is standardized to regional reference conditions, and the biological criteria threshold is often established as a percentile of the distribution of reference site scores. The index is the basis for numeric biological criteria in many States and Tribes (U.S. EPA 2002a).

The alternative to a single broad use is to divide the continuum of biological condition (the BCG) into several tiers for more precise management. As mentioned earlier, tiered aquatic life uses couple narrative descriptions of the use with criteria for measuring attainment of the use. Ideally, the narrative descriptions should incorporate biologically meaningful differences among tiers. The BCG provides an interpretative framework for defining reference conditions and articulating the biological condition that is being protected or restored in the water of interest.

Several States and Tribes have adopted tiered aquatic life use statements in their water quality standards and some are developing the technical program and further tightening the linkage between their narrative use statements and numeric biological criteria (U.S. EPA 2002a). For example, Texas has had tiered aquatic life uses identified in their water quality standards for surface waters since 1984 (Table 1-1). Texas' current WQS identify numeric dissolved oxygen criteria and include narrative aquatic life attributes. Numeric biological criteria have been developed for assessing both fish and benthic macroinvertebrate communities in wadeable streams. If site-specific conditions do not meet criteria for "High" use category as determined by receiving water assessment, a use attainability analysis will be conducted. Texas continues to evaluate the application of biological criteria for other aquatic systems, but at this point does not have a specific action plan to adopt numeric biological criteria for those systems. Other States cited elsewhere in this document, e.g., Maine, New Jersey, Ohio, and Vermont, have either developed or are considering developing tiered aquatic life uses. Though these approaches for tiering aquatic life uses may differ in detail and assessment methods, their uses share the same core elements:

- Biological information is the basis for the use designations.
- Numeric biological indicators or biocriteria are developed for each use.
- Development of tiers based on data from comprehensive, robust monitoring program.

TABLE 1-1. Aquatic Life Subcategories in Texas WQS (Figure: 30 TAC §307.7(b)(3)(A)(i)).

Aquatic Life Use Subcategory	Dissolved Oxygen Criteria, mg/L			Aquatic Life Attributes					
	Freshwater mean/ minimum	Freshwater in Spring mean/ minimum	Saltwater mean/ minimum	Habitat Character-istics	Species Assemblage	Sensitive species	Diversity	Species Richness	Trophic Structure
Exceptional	6.0/4.0	6.0/5.0	5.0/4.0	Outstanding natural variability	Exceptional or unusual	Abundant	Exceptionally high	Exceptionally high	Balanced
High	5.0/3.0	5.5/4.5	4.0/3.0	Highly diverse	Usual association of regionally expected species	Present	High	High	Balanced to slightly imbalanced
Intermediate	4.0/3.0	5.0/4.0	3.0/2.0	Moderately diverse	Some expected species	Very low in abundance	Moderate	Moderate	Moderately imbalanced
Limited	3.0/2.0	4.0/3.0		Uniform	Most regionally expected species absent	Absent	Low	Low	Severely imbalanced

- Dissolved oxygen means are applied as a minimum average over a 24-hour period.
- Daily minima are not to extend beyond 8 hours per 24-hour day. Lower dissolved oxygen minima may apply on a site-specific basis, when natural daily fluctuations below the mean are greater than the difference between the mean and minima of the appropriate criteria.
- Spring criteria to protect fish spawning periods are applied during that portion of the first half of the year when water temperatures are 63.0°F to 73.0°F.
- Quantitative criteria to support aquatic life attributes are described in the standards implementation procedures.
- Dissolved oxygen analyses and computer models to establish effluent limits for permitted discharges will normally be applied to mean criteria at steady-state, critical conditions.
- Determination of standards attainment for dissolved oxygen criteria is specified in §307.9(d)(6) (relating to Determination of Standards Attainment).

The insights and experiences from States and Tribes that have adopted tiered aquatic life uses and numeric biocriteria in their water quality standards, as well as from those currently developing biological assessment and criteria programs, reveal the values of tiered aquatic life uses implemented in State and Tribal WQS (Table 1-2).

TABLE 1-2. The benefits and WQS regulation context for TALUs.

Value-added	Explanation	Supporting WQS Regulation
Set more appropriate designated ALUs	Define ALUs in a more precise way that is neither under-protective of existing high-quality resources nor overprotective for waters that have been extensively and irretrievably altered	40CFR131.10 40CFR131.12 (Protect High Quality Waters) 40CFR130.23 (Support attainment decisions and diagnose causes)
Strengthen the linkage between designated ALUs and how attainment is assessed	TALUs help to clarify and refine water quality goal statements so numeric biological, chemical and physical criteria can be adopted to protect the use	40CFR131.10(c) 40CFR131.12 (Protect High Quality Waters) 40CFR130.23 (Support attainment decisions and diagnose causes)
Enhance public understanding and participation in setting water quality goals	TALUs provide a common frame of reference or generic yardstick to more clearly recognize common ground and differences in desired environmental goals of various stakeholders as designated uses are adopted	40 CFR131.20 (a)(b)

Building on these "lessons learned," the U.S. EPA convened a technical workgroup in 2000 to identify existing scientifically sound and practical approaches that would help States and Tribes provide more precision, or specificity, in their designated aquatic life uses. The workgroup included biologists and aquatic ecologists from States, Tribes, U.S. EPA, USGS, the academic research community, and the private sector. The workgroup was asked to address the following questions:

- What are effective technical approaches using biological information to provide more specificity in their designated aquatic life uses?

- What are the "lessons learned" that can be capitalized on and shared with other States and Tribes?

The workgroup was charged with developing a scientific framework using biological information to better define designated aquatic life uses, enabling more precise use descriptions. Their product is a narrative model describing graduated tiers of biological response to increasing levels of stressors, the *Biological Condition Gradient (BCG).* The model is founded on peer-reviewed work in the field of bioassessments over the past thirty years (Fausch et al. 1984, Karr et al. 1986, Cairns and Pratt 1993, Barbour et al. 1999) and on the experiences and empirical observations of States and Tribes that have developed tiered aquatic life uses and biological criteria for use in their water programs (Courtemanch et al. 1989, Courtemanch 1995, Yoder 1995, Yoder and Rankin 1995b).

1.5 The Biological Condition Gradient: A tool for better defining and developing more precise aquatic life uses

The Biological Condition Gradient (BCG) is a scientific model for interpreting biological response to increasing effects of stressors on aquatic ecosystems (Figure 1-1). The model describes how ten attributes of aquatic ecosystems change in response to the increasing levels of stressors. The attributes include several aspects of community structure, organism condition, ecosystem function, and spatial and temporal attributes of stream size and connectivity. The gradient can be considered analogous to a field-based dose-response curve where dose (x-axis) = increasing levels of stressors and response (y-axis) = biological condition (see figure below). The BCG differs from the standard dose-response curve, in that the BCG does not represent the laboratory response of a single species to a specified dose of a known chemical, but rather the *in-situ* response of the biota to the sum of stresses it is exposed to. The BCG is divided into six tiers of biological condition along the stressor-response curve, ranging from observable biological conditions found at no or low levels (Tier 1) to those found at high levels of stressors (Tier 6). The BCG model was developed to provide a common framework for interpreting biological information regardless of methodology and geography. When calibrated to a regional or state scale, States and Tribes can use the model to more precisely evaluate the current and potential biological condition of their waters and use that information to better define their aquatic life uses. Additionally, States and Tribes can use this interpretative model to more clearly communicate the condition of their aquatic resources to the public.

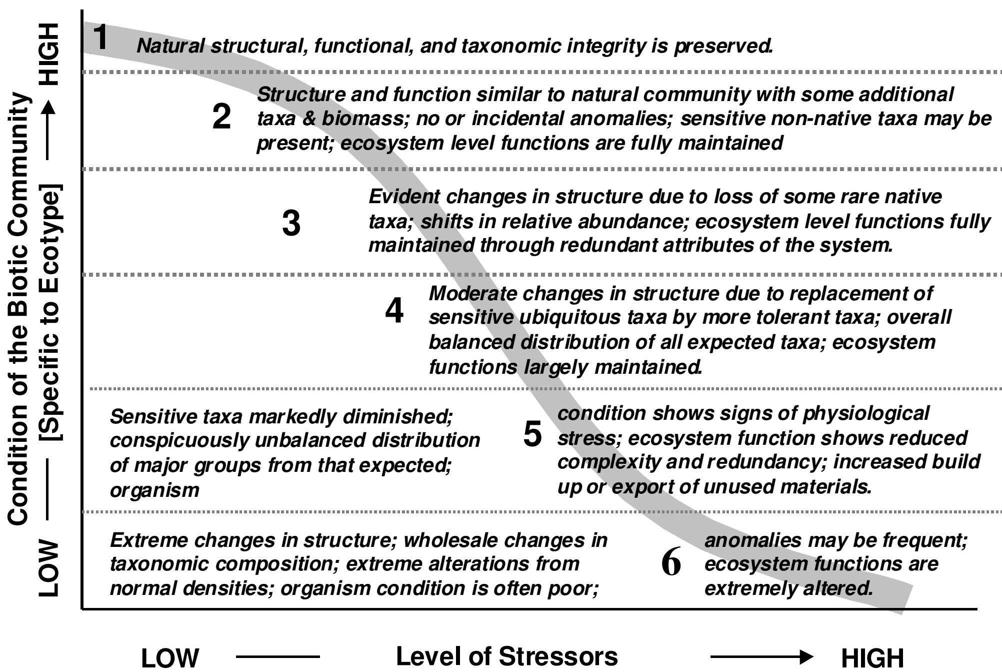


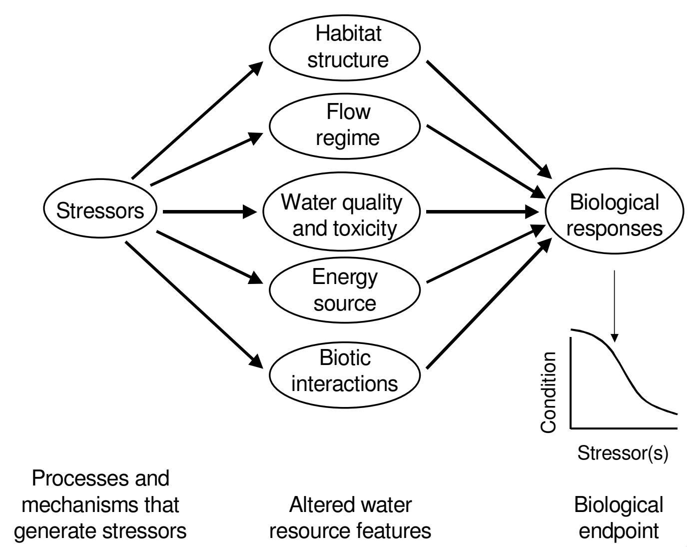
FIGURE 1-1. Conceptual model of the Biological Condition Gradient.

The BCG model was developed based on common patterns of biological response to stressors observed empirically by aquatic biologists and ecologists from different geographic areas of the U.S. Once a draft model was constructed, it was tested at a workgroup meeting and then at two regional workshops. The model was tested by determining how consistently the scientists assigned samples of macroinvertebrates or fish to the different tiers of biological condition. Workgroup members identified similar sequences of biological response to increasing levels of stressors regardless of geographic area. These results support the use of the BCG as a nationally applicable model for interpreting the biological condition of aquatic systems. Chapter 2 discusses the development and makeup of the conceptual BCG and Chapter 3 explores strategies for regionally modifying, or calibrating, the conceptual model. Chapter 4 describes how the x-axis of the BCG model, the stressor gradient, can be characterized and explains how the effects of stressors on biological condition play a role in constructing and using a BCG. Chapter 5 discusses the underlying principles and processes States have learned in using biological information to develop tiered aquatic life uses, and examples of how States have applied tiered uses in water quality management are presented in Chapter 6.

Integral to the development of the BCG is characterizing the model's x-axis, the stressor gradient (Figure 1-1). **Stressors** are physical, chemical or biological factors that induce an adverse response from aquatic biota (U.S. EPA 2000b; EPA/822/B-00/025). For example, high concentrations of certain metals, nutrients, or sediment can adversely impact aquatic biota. Loss of aquatic habitat or presence of aquatic invasive species can also adversely impact, or stress, the aquatic biota expected for a specific waterbody. These stressors can cause aquatic ecosystems to change from natural conditions, exhibiting altered compositional, structural, and functional characteristics. The degree to which stressors affect the biota depends on the magnitude, frequency, and duration of the exposure of the biota to the stressors. Developing a BCG for a given system characterizes the general relationship between its stressors in total (the model's x-axis) and its overall biological condition (the y-axis). Multiple stressors are usually present, and thus the stressor x-axis of the BCG seeks to represent their cumulative influence as a **Generalized Stressor Gradient** (GSG), much as the y-axis generalizes biological condition.

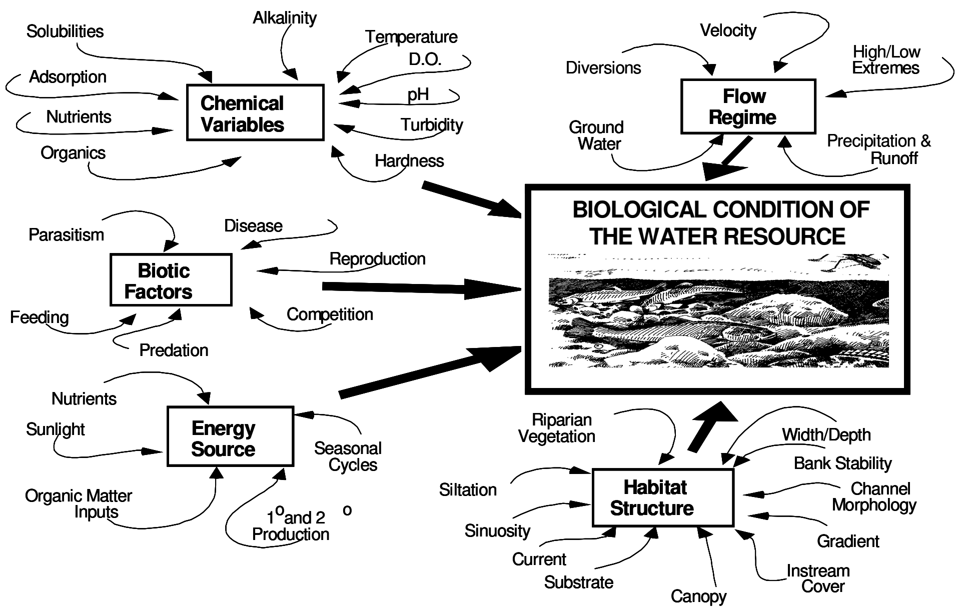
Understanding the links between stressors and their sources and the response of the aquatic biota will help to more accurately determine the existing and potential condition of the aquatic biota (Figure 1-2). There are different approaches and emerging science to define and quantify the causal sequence between stressors and their sources and biological responses. Building on current State and Tribal approaches, a framework for characterizing stressors, the processes and mechanisms that generate them, and the resulting biological response is presented. This framework may not only help State and Tribal managers more precisely define designated uses, including potential future uses, but may support diagnosis of use impairment and help prioritize management decision making.

FIGURE 1-2. The causal sequence from stressors and their sources through the five major water resource features to the biological responses, i.e., the biological endpoints. This model illustrates the multiple pathways that stressors and their sources can affect aquatic biota. Insert illustrates the relationship between stressor dose and the gradient of biological responses (after Karr and Yoder 2004; used by permission of J.D. Allan, originally presented at the 2002 TALU Workgroup Meeting).



1.6 Conceptual basis for the Biological Condition Gradient

The five factors that determine the integrity of a water resource, which were originally described by Karr and Dudley (1981; Figure 1-3), have been consistently used as the conceptual basis for biological assessment and tiered aquatic life uses. In the context of the TALU approach, consideration of the five factors in Figure 1-3 are components of the stressor axis of the BCG model, while the condition of the water resource is accounted for by the response of the biological community to the stressors, the Biological Condition Gradient (BCG). The health and well-being of the aquatic biota is an important barometer to measure progress towards achieving Clean Water Act goals. Biological integrity has been defined as the combined result of chemical, physical, and biological processes in the aquatic environment (Karr and Dudley 1981, Karr et al. 1986). Biological criteria help reconcile the mosaic of factors and interactions that exist, parts of which may be characterized and measured using chemical and physical indicators.



(Modified from Karr et al. 1986).

FIGURE 1-3. The five major factors that determine the biological condition of aquatic resources (modified from Karr et al. 1986).

An important conceptual foundation of tiered aquatic life uses is the "position of the standard" that was described by the National Research Council Committee on Science in TMDLs (NRC 2001; Figure F-1). This concept describes the "position" of different types of criteria with respect to their position along a causal chain of indicators beginning with sources (stressor indicators), to changes in pollutant contributions or attributes of landscape and/or hydrology that emanate from those sources (exposure indicators), to instream exposures (pollutants, attributes of habitat), to indicators of biological condition (response indicators) that directly assess the designated use. Because designated uses are written in qualitative, narrative terminology, the challenge is to relate a criterion to the designated use. In general, establishing this relationship becomes easier as the criterion is positioned closer to the designated use, hence the NRC recommendation on the use of biological information to help determine more appropriate aquatic life uses and to couple the narrative use statements with quantitative methods. Thus biological criteria can fill a gap along this position spectrum and serve a useful role in the expression and implementation of water quality standards.

Karr and Yoder (2004) further elaborated upon this concept by adding the interactive relationships between pollution and pollutants from both point and nonpoint sources (Figure 1-4). It also relates different types of indicators in the causal sequence of events and exemplifies the appropriate roles of chemical, physical, and biological parameters as stressor, exposure, and response indicators (Yoder and Rankin 1998). In this scheme, attainment of a designated use is the desired result of the management of stressors (chemical, biological, physical) and is explained by how stressors influence and change the five factors that determine the integrity of an aquatic resource (Karr and Yoder 2004). In each of these process descriptions, the end outcome of water quality management is reflected in the status of a designated use. Attainment of the designated use confirms the effectiveness of the sequence of management actions; non-attainment is evidence of an incomplete process and a prompt to re-examine the management strategy. Each provides important feedback about the effectiveness of management strategies. Therefore, how designated uses are developed, assigned, and measured is key to the outcomes derived from water quality management.

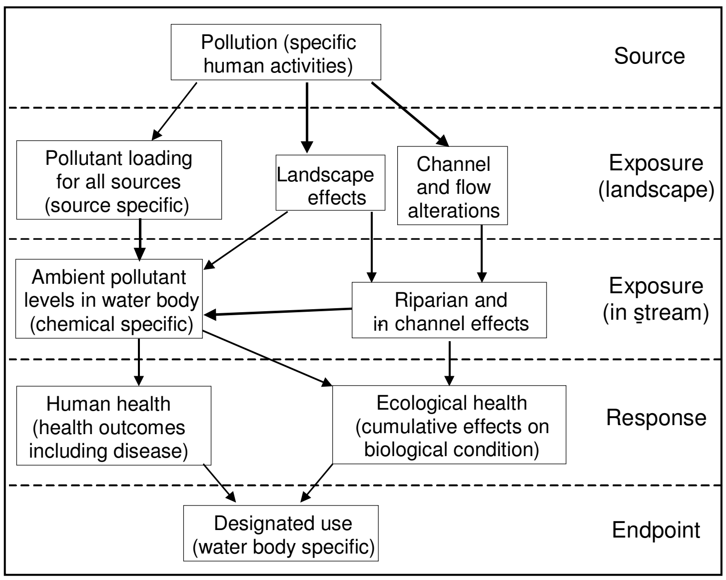


FIGURE 1-4. Modification of the NRC "position of the criterion" concept (Figure F-1) showing the causal sequence from *indicators* of stress, exposure, and response in relation to point and nonpoint source impacts, specific types of criteria, and designated uses that define the endpoints of interest to society (after Karr and Yoder 2004).

1.7 Key points from Chapter 1

1. Section 101(a) of the CWA establishes broad national goals and objectives such as the chemical, physical, and biological integrity objective. To help achieve the integrity objective, the CWA also established, among other things, an interim goal for the protection and propagation of fish, shellfish, and wildlife. The protection and propagation interim goal has been interpreted by U.S. EPA to include the protection of the full complement of aquatic organisms residing or migrating through a waterbody. The health and well-being, or condition, of the aquatic biota is an important barometer to measure progress towards achieving Clean Water Act goals and objectives.

2. State water quality standards provide the foundation for water quality-based pollution control programs. With the public participating in their adoption (see 40 CFR 131.20), such standards serve the dual purposes of establishing the water quality goals for a specific waterbody *(designated uses)* and serve as the regulatory basis for the establishment of water quality-based

treatment controls and strategies beyond the technology-based levels of treatment required by Sections 301(b) and 306 of the CWA.

3. A waterbody's *designated use(s)* are those uses specified in water quality standards, whether or not they are being attained (40 CFR 131.3(f)). The "use" of a waterbody is the most fundamental articulation of its role in the aquatic and human environments. All of the water quality protections established by the CWA follow from the waterbody's designated use.

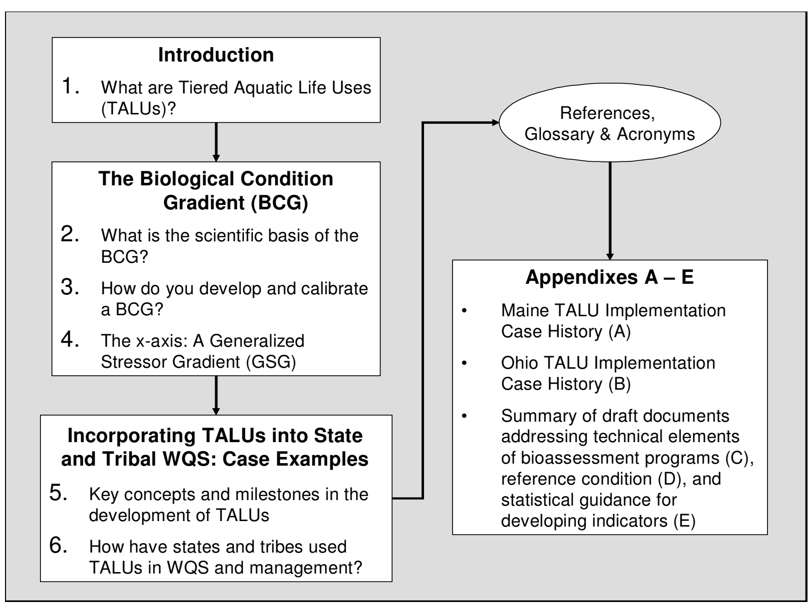
4. *Tiered aquatic life uses* are bioassessment-based statements of expected biological condition in specific waterbodies. Tiered uses allow more precise and measurable definitions of *designated aquatic life uses*.

5. Several States and Tribes have adopted tiered aquatic life uses in their water quality standards. This document is based on the "lessons learned" from their experiences and the recommendations from a technical workgroup charged with integrating existing scientifically sound and practical approaches to 1) tier designated aquatic life uses using biological information, and 2) incorporate information on sources of stress as drivers of biological condition.

1.8 Organization of the document

This chapter provided the background and rationale for using biological information to designate aquatic life uses in tiers that more specifically differentiate the characteristics of the biological community currently present or desired in a waterbody. The following chapters are based on the recommendations of the TALU technical workgroup tasked with identifying existing scientifically sound and practical approaches that would help States and Tribes provide more precision, or specificity, in their designated aquatic life uses (Figure 1-5). Chapters 2 and 3 discuss the Biological Condition Gradient (BCG) – what it is, how the national conceptual model was developed and tested, and how to calibrate the conceptual model to a region. Chapter 4 describes how the x-axis of the BCG model, the stressor gradient, can be characterized and explains how the effects of stressors on biological condition play a role in constructing and using a BCG. Chapter 5 provides examples on how States have developed tiered aquatic life uses. The experiences of Maine and Ohio, two States that have completed this process, serve as comprehensive case histories that are found in Appendixes A and B. Chapter 6 details how Maine and Ohio have used tiered aquatic life uses in assessment and management as examples that might guide future implementation guidance.

FIGURE 1-5. Roadmap to the document.



Related Technical Support Documents:

Appendixes C, D & E contain summaries of three "companion" documents that are under development. Each contains detailed information relevant to developing tiered aquatic life uses, including components of State and Tribal bioassessment programs, statistical methods that use biological data, and best practices for developing reference conditions. Following is a brief description of each document.

Technical Guidelines: Technical Elements of a Bioassessment Program – DRAFT

This document is intended primarily for use by State and Tribal program managers and staff who are responsible for monitoring and assessment and water quality standards programs. The document describes the technical attributes of biological assessment programs, and can thus be used by States and Tribes to 1) determine where they are in the biological assessment and criteria development processes, and 2) develop, structure, and, if necessary, modify their programs and refine designated aquatic life uses.

U.S. EPA project leads: Susan Jackson, Office of Water; Ed Hammer, Region 5; Tina Laidlaw, Region 8; and Gretchen Hayslip, Region 10

The Role of Reference Condition in Biological Assessment and Criteria – DRAFT DOCUMENT ON DEVELOPMENT AND APPLICATION OF THE REFERENCE CONDITION CONCEPT

This document will provide States, Tribes, and other practitioners with guidelines on using reference conditions in their water management programs, particularly for ecological assessments. The guidelines described are intended to facilitate greater implementation of best practices for reference condition, thereby improving the success of individual programs and leading to greater consistency among States and Tribes.

U.S. EPA project leads: Evan Hornig, Office of Water; Phil Larsen, Office of Research and Development; and Wayne Davis, Office of Environmental Information

Statistical Guidance for Developing Indicators for Rivers and Streams: A Guide for Constructing Multimetric and Multivariate Predictive Bioassessment Models – DRAFT

This document will provide methods and outlines the steps required to complete multimetric and multivariate predictive assessment models, two methods for analyzing and assessing waterbody condition from assemblage and community-level biological information.

U.S. EPA project lead: Florence Fulk, Office of Research and Development

The Biological Condition Gradient

The Biological Condition Gradient (BCG) is a scientific model that allows consistent interpretation of biological condition although assessment approaches may differ. The BCG combines scientific knowledge with the practical experience and needs of resource managers and can assist environmental practitioners in the U.S. to better:

- define aquatic resources
- establish direct relationships between biological condition and stressors
- communicate clearly to the public both the existing and potential uses of a waterbody

The BCG is consistent with ecological theory and is a means for standardizing interpretations of the response of aquatic biota to stressors. The model should facilitate communication among scientists, managers, and the public on the current conditions and ecological potential for specific waterbodies.

Chapter 2 outlines the development and makeup of the BCG model. The BCG describes changes in ten ecological attributes across a gradient of biological condition caused by increasing stressors (Table 2-1). It is divided into six condition tiers, Tier 1 representing natural, or undisturbed, conditions through Tier 6 representing severely altered conditions.

TALU Workgroup biologists from across the U.S. agreed that a similar sequence of biological alterations occur in streams in response to stressors, strengthening the feasibility of using the BCG as a common framework to guide management decisions that protect and restore aquatic systems in the U.S. (Davies and Jackson in press). The model is consistent with ecological theory and can be adapted or calibrated to reflect specific geographic regions. Scientific knowledge can be reviewed and consolidated and research needs can be expressed in a context relevant to management. Thus, the model also serves as a framework that 1) synthesizes what has been observed into testable hypotheses, and 2) identifies knowledge gaps in need of further research.

Chapter 3 explores strategies for regionally modifying, or calibrating, the BCG including approaches for recalibrating existing indexes. Three States (Maine, Ohio, and Vermont) have incorporated a BCG into their water quality standards as well as numeric criteria. Several other States (e.g., New Jersey, Texas, and a consortium of New England states) have begun the process of evaluating the potential use of a BCG. Each of these States is following basically the same approach used by the national TALU Workgroup to develop the BCG model, reaching consensus among regional biological experts familiar with natural aquatic communities and their responses to stress.

Chapter 4 describes the model's x-axis, the stressor gradient that illustrates alteration in biological condition. The degree to which stressors affect the biota depends on the magnitude, frequency, and duration of the exposure of the biota to the stressors. Developing a BCG for a given system characterizes the general relationship between its stressors in total (the model's x-axis) and its overall biological condition (the y-axis). Multiple stressors are usually present, and thus the stressor x-axis of the BCG seeks to represent their cumulative influence as a **Generalized Stressor Gradient** (GSG), much as the y-axis generalizes biological condition. Chapter 4 explains how stressors can be characterized and describes how the influence of stressors on biological condition plays a role in constructing and using a BCG.

CHAPTER 2. WHAT IS THE SCIENTIFIC BASIS OF THE BIOLOGICAL CONDITION GRADIENT?

The Biological Condition Gradient (BCG) extends the empirical work of earlier researchers and practitioners to create a nationally consistent model that links management goals for resource condition with the quantitative measures used in biological assessments. The BCG was designed to describe ecological response to stressors in sufficient detail so that a site can be placed into a tier along the BCG continuum through use of the core data elements collected by most State or Tribal monitoring programs.

The practice of using biological indicators to assess water quality is over a century old. The Saprobien System, a concept proposed by Lauterborn in 1901and further developed the following year by Kolkwitz and Marsson (Davis 1995), uses benthic macroinvertebrates and planktonic plants and animals as indicators of organic loading and low dissolved oxygen, and has been updated and is currently used in several European countries. Concurrently, the limnologists Thienemann and Naumann developed the concept of trophic state classification for lakes in the 1920s (Carlson 1992, Cairns and Pratt 1993). These early indexes described a response gradient (or response classes for lakes) to enrichment. The Saprobien System was explicitly developed to assess human pollution in rivers, but the trophic state concept was originally developed to describe natural conditions in lakes and only later became a concept to describe pollution-caused eutrophication (e.g., Vollenweider 1968). The 1950s marked the development of Beck's biotic index in the U.S. and Pantle and Buck's Saprobic Index in Europe, which were directly based on the Saprobien System (cited in Davis 1995). The Saprobic Index, which led to the development of the widely used Hilsenhoff Index (e.g., Hilsenhoff 1987) in the U.S., could be considered the predecessor of today's biotic indexes (Davis 1995).

The conceptual foundation of the BCG is based on many decades of biologists' accumulated experience with biological assessment and monitoring. Biological information from monitoring programs has been frequently synthesized by constructing biotic indexes, such as the Index of Biotic Integrity (IBI) (Karr 1981, Karr et al. 1986). The IBI integrated the concept of anchoring the measurement system in undisturbed reference conditions with the measurement of several indicators intended to reflect ecological components of composition, diversity, and ecosystem processes. It thus combined a conceptual model of ecosystem change in response to increasing levels of stressors with a practical measurement system for fish. The BCG is also grounded in the concepts in Cairns et al. (1993) describing "natural" conditions and the change in biological condition caused by stressors. To achieve maximum potential application nationwide, the BCG tiers were developed based on States' various experiences designing and implementing tiered aquatic life use and management goals as well as the practical experience of aquatic scientists from different bio-geographic areas, each of whom had fifteen to thirty years of experience in the field. The BCG:

1. Describes a complete scale of condition from natural (Tier 1) to severely altered (Tier 6);
2. Synthesizes existing field observations and generally accepted interpretations of patterns of biological change within a common framework; and
3. Helps determine the degree to which a system may have departed from natural condition, based on measurable, ecologically important attributes.

At present, the description of biological attributes that make up the model applies best to permanent, hard-bottom streams that are exposed to increases in temperature, nutrients, and fine sediments because this is the stream-type and stressor regime originally described by the model. The model has been further tested with States and Tribes in different parts of the country (e.g., arid west and great plains) to evaluate the national applicability of the model. Results have been successful with some necessary refinement of the model attributes to accommodate regional differences. For example, during a workshop in Texas where the BCG was being evaluated using Texas data, Attribute II (sensitive-rare taxa) was redefined as *highly sensitive taxa* because rarity of a taxon in the region was not deemed to be associated with sensitivity to stress. In arid streams, many rare, native taxa are highly tolerant to stressors such as low dissolved oxygen and high temperature. Thus, the BCG can be applicable to other aquatic ecosystems and stressors with appropriate modifications. The BCG should be viewed as an evolving model that must be responsive to changes in scientific understanding resulting from the analysis of empirical data.

The value of a heuristic model such as the BCG is not only that it documents experimentally established knowledge, but also that it promotes a more rigorous testing of empirical observations by clearly stating them in a provisional model. Conceptual models formalize the state of knowledge and guide research. Empirically based generalizations have led to conceptual models that describe the behavior of biological systems under stress (Brinkhurst 1993; Margalef 1963, 1981; Odum, et al. 1979; Rapport et al. 1985; Schindler 1987; Fausch et al. 1990; Karr and Dudley 1981). For example, Brinkhurst observed that "Everyone knew [in 1929] that increases in numbers and species could be related to mild pollution, that moderate pollution could produce changes in taxa so that diversity remained similar but species composition shifted, and that eventually species richness declined abruptly and numbers of some tolerant forms increased dramatically." Such ecosystem responses to stressor gradients have been portrayed as a progression of stages that occur in a generally consistent pattern (Odum et al. 1979, Odum 1985, Rapport et al. 1985, Cairns and Pratt 1993). Establishing and validating quantifiable thresholds along that progression with empirical data is a priority need for resource managers (Cairns l981).

2.1 What the BCG model looks like

The BCG model depicts ecological condition in terms of ten system attributes expressed at different spatial scales (Table 2-1). In biological assessments, most information is collected at the spatial scale of a site or reach and the temporal scale of a single sampling event. Many of the attributes that make up the BCG are based on these scales. Site scale attributes include aspects of taxonomic composition and community structure (Attributes I-VI) and organism and system performance (Attributes VII and VIII). At larger temporal and spatial scales, physical-biotic interactions (Attributes IX and X) were also included because of their importance in evaluating the longer term impacts, restoration potential and recoveries.

TABLE 2-1. Biological Condition Gradient matrix.

Ecological Attributes	Biological Condition Gradient Tiers					
	1 Natural or native condition	2 Minimal changes in the structure of the biotic community and minimal changes in ecosystem function	3 Evident changes in structure of the biotic community and minimal changes in ecosystem function	4 Moderate changes in structure of the biotic community and minimal changes in ecosystem function	5 Major changes in structure of the biotic community and moderate changes in ecosystem function	6 Severe changes in structure of the biotic community and major loss of ecosystem function
	Native structural, functional and taxonomic integrity is preserved; ecosystem function is preserved within the range of natural variability	Virtually all native taxa are maintained with some changes in biomass and/or abundance; ecosystem functions are fully maintained within the range of natural variability	Some changes in structure due to loss of some rare native taxa; shifts in relative abundance of taxa but Sensitive-ubiquitous taxa are common and abundant; ecosystem functions are fully maintained through redundant attributes of the system	Moderate changes in structure due to replacement of some Sensitive-ubiquitous taxa by more tolerant taxa, but reproducing populations of some Sensitive taxa are maintained; overall balanced distribution of all expected major groups; ecosystem functions largely maintained through redundant attributes	Sensitive taxa are markedly diminished; conspicuously unbalanced distribution of major groups from that expected; organism condition shows signs of physiological stress; system function shows reduced complexity and redundancy; increased build-up or export of unused materials	Extreme changes in structure; wholesale changes in taxonomic composition; extreme alterations from normal densities and distributions; organism condition is often poor; ecosystem functions are severely altered
I Historically documented, sensitive, long-lived or regionally endemic taxa	As predicted for natural occurrence except for global extinctions	As predicted for natural occurrence except for global extinctions	Some may be absent due to global extinction or local extirpation	Some may be absent due to global, regional or local extirpation	Usually absent	Absent
II Sensitive-rare taxa	As predicted for natural occurrence, with at most minor changes from natural densities	Virtually all are maintained with some changes in densities	Some loss, with replacement by functionally equivalent Sensitive-ubiquitous taxa	May be markedly diminished	Absent	Absent
III Sensitive-ubiquitous taxa	As predicted for natural occurrence, with at most minor changes from natural densities	Present and may be increasingly abundant	Common and abundant; relative abundance greater than Sensitive-rare, taxa	Present with reproducing populations maintained; some replacement by functionally equivalent taxa of intermediate tolerance.	Frequently absent or markedly diminished	Absent
IV Taxa of intermediate tolerance	As predicted for natural occurrence, with at most minor changes from natural densities	As naturally present with slight increases in abundance	Often evident increases in abundance	Common and often abundant; relative abundance may be greater than Sensitive-ubiquitous taxa	Often exhibit excessive dominance	May occur in extremely high OR extremely low densities; richness of all taxa is low
V Tolerant taxa	As naturally occur, with at most minor changes from natural densities	As naturally present with slight increases in abundance	May be increases in abundance of functionally diverse tolerant taxa	May be common but do not exhibit significant dominance	Often occur in high densities and may be dominant	Usually comprise the majority of the assemblage; often extreme departures from normal densities (high or low)

TABLE 2-1. Biological Condition Gradient matrix.

	Biological Condition Gradient Tiers					
	1 Natural or native condition	2 Minimal changes in the structure of the biotic community and minimal changes in ecosystem function	3 Evident changes in structure of the biotic community and minimal changes in ecosystem function	4 Moderate changes in structure of the biotic community and minimal changes in ecosystem function	5 Major changes in structure of the biotic community and moderate changes in ecosystem function	6 Severe changes in structure of the biotic community and major loss of ecosystem function
VI Non-native or intentionally introduced taxa	Non-native taxa, if present, do not displace native taxa or alter native structural or functional integrity	Non-native taxa may be present, but occurrence has a non-detrimental effect on native taxa	Sensitive or intentionally introduced non-native taxa may dominate some assemblages (e.g. fish or macrophytes)	Some replacement of sensitive non-native taxa with functionally diverse assemblage of non-native taxa of intermediate tolerance	Some assemblages (e.g., fish or macrophytes) are dominated by tolerant non-native taxa	Often dominant; may be the only representative of some assemblages (e.g., plants, fish, bivalves)
VII Organism Condition (especially of long-lived organisms)	Any anomalies are consistent with naturally occurring incidence and characteristics	Any anomalies are consistent with naturally occurring incidence and characteristics	Anomalies are infrequent	Incidence of anomalies may be slightly higher than expected	Biomass may be reduced; anomalies increasingly common	Long-lived taxa may be absent; Biomass reduced; anomalies common and serious; minimal reproduction except for extremely tolerant groups
VIII Ecosystem Functions	All are maintained within the natural range of variability	All are maintained within the natural range of variability	Virtually all are maintained through functionally redundant system attributes; minimal increase in export except at high storm flows	Virtually all are maintained through functionally redundant system attributes though there is evidence of loss of efficiency (e.g., increased export or decreased import)	There is apparent loss of some ecosystem functions manifested as increased export or decreased import of some resources, and changes in energy exchange rates (e.g., P/R; decomposition)	Most functions show extensive and persistent disruption
IX Spatial and temporal extent of detrimental effects	N/A A natural disturbance regime is maintained	Limited to small pockets and short duration	Limited to the reach scale and/or limited to within a season	Mild detrimental effects may be detectable beyond the reach scale and may include more than one season	Detrimental effects extend far beyond the reach scale leaving only a few islands of adequate conditions; effect extends across multiple seasons	Detrimental effects may eliminate all refugia and colonization sources within the catchment and affect multiple seasons
X Ecosystem connectance	System is highly connected in space and time, at least annually	Ecosystem connectance is not impacted	Slight loss of connectance but there are adequate local recolonization sources	Some loss of connectance but colonization sources and refugia exist within the catchment	Significant loss of ecosystem connectance is evident; recolonization sources do not exist for some taxa	Complete loss of ecosystem connectance in at least one dimension (i.e., longitudinal, lateral, vertical, or temporal) lowers reproductive success of most groups; frequent failures in reproduction & recruitment

2.1.1 The BCG Attributes

Taxonomic Composition and Structure: Attributes I – VI

Attribute I: Historically documented, sensitive, long-lived or regionally endemic taxa.
"Historically documented" refers to taxa known to have been supported in a waterbody or region prior to enactment of the 1972 Clean Water Act, according to historical records compiled by State or federal agencies or published scientific literature.

"Sensitive or regionally endemic taxa" have restricted, geographically isolated distribution patterns (occurring only in a locale as opposed to a region), often due to unique life history requirements. They may be long-lived, late maturing, low fecundity, limited mobility, or require a mutualist relation with other species. They may be among listed Endangered or Threatened (E/T) or special concern species. Predictability of occurrence is often low, and therefore requires documented observation. Recorded occurrence may be highly dependent on sample methods, site selection, and level of effort.

Attribute II: Sensitive-rare taxa.
These are taxa that naturally occur in low numbers relative to total population density but may make up a large relative proportion of richness. They may be ubiquitous in occurrence or restricted to certain micro-habitats, but because of low density, recorded occurrence is dependent on sample effort. Often stenothermic (having a narrow range of thermal tolerance) or cold-water obligates; commonly k-strategists (populations maintained at a fairly constant level; slower development; longer life-span). May have specialized food resource needs or feeding strategies. Generally intolerant to significant alteration of the physical or chemical environment; are often the first taxa observed to be lost from a community.

Attribute III: Sensitive ubiquitous taxa.
"Sensitive" taxa from Attributes II and III are taxa that are intolerant to a given stress; they are the first species affected by the specific stressor to which they are "sensitive" and the last to recover following restoration. Sensitive ubiquitous taxa are ordinarily common and abundant in natural communities when conventional sampling methods are used. They often have a broader range of thermal tolerance than Sensitive-rare taxa and comprise a substantial portion of natural communities and often exhibit negative response (loss of population, richness) at mild pollution loads or habitat alteration.

Attribute IV: Taxa of intermediate tolerance.
Taxa that comprise a substantial portion of natural communities; may be r-strategists (early colonizers with rapid turn-over times; e.g.," boom/bust population characteristics). May be eurythermal (having a broad thermal tolerance range). May have generalist or facultative feeding strategies enabling utilization of relatively more diversified food types. Readily collected with conventional sample methods. May increase in number in waters with moderately increased organic resources and reduced competition but are intolerant of excessive pollution loads or habitat alteration.

Attribute V: Tolerant taxa.
Taxa that comprise a low proportion of natural communities. Taxa often are tolerant of a broader range of environmental conditions and are thus resistant to a variety of pollution or habitat induced stress. They may increase in number (sometimes greatly) in the absence of competition. Commonly r-strategists (early colonizers with rapid turn-over times: e.g., "boom/bust" population characteristics), able to capitalize when stress conditions occur. These taxa are the last survivors in highly disturbed systems.

***Taxa tolerance to stressors* (ATTRIBUTES I-V).**
Taxa differ in their sensitivities to stressors. Changes in the numbers, kinds and relative abundance of taxa across stressor gradients are important and useful indicators of adverse

effects (Cairns 1977, Karr 1981). Sensitivity of taxa to stress can vary among species, as well as with stressor. Shifts in taxa as a function of differing sensitivities to aquatic and riparian disturbance are well documented (Table 2-2). For perennial streams in temperate zones, disturbance tends to select for short-lived, tolerant species and against longer-lived, less tolerant species (Pianka 1970, Odum 1985, Rapport et al. 1985). In the highest quality tiers of the BCG, locally endemic taxa that are long-lived and ecologically specialized are well represented. With increasing stress, assemblage composition shifts towards tolerant species or short-lived taxa that can rapidly colonize disturbed environments. Assemblages in the lower tiers are dominated by eurytopic taxa (those with wide environmental ranges) with generalist or facultative feeding strategies.

TABLE 2-2. Evidence in support of the depicted changes in ecological attributes in the BCG.

BCG Attribute	Response	Case-specific documentation	Reference
I-V	Shifts in the numbers and kinds of species present, and in the number of individuals per species, as a function of varying tolerances to different kinds of aquatic and riparian disturbance.	changes in lake diatom species composition in response to intentional fertilization	Zeeb et al. 1974; Yang et al. 1996
		loss of sculpins downstream of metal mines	Mebane et al. 2003
		changes in algal species across a nutrient gradient in the Florida Everglades	Stevenson et al. 2002
		changes in diatom assemblages with increased acidification and eutrophication of lakes	Dixit et al. 1999
		shifts in species composition along a gradient of pulp and paper mill effluent concentration in a Maine river	Rabeni et al. 1988
		shifts in damselfly species from specialist species to generalist species along a gradient of organic pollution in an Italian river	Solimini et al. 1997
		variable sensitivities of benthic macroinvertebrate species to acidic conditions	Courtney and Clements 2000
		changes in fish species composition in an Oregon river with increased nutrients and temperature	Hughes and Gammon 1987
		differentially tolerant fish species in response to heavy metal and dissolved oxygen gradients in two Indian rivers	Ganasan and Hughes 1998
		decline in darters, sunfish, and suckers as well as other intolerant fishes and increase in tolerant fishes in the Midwest	Karr et al 1986; Yoder and DeShon 2003
		variable responses of stream amphibians to severe siltation	Welsh and Ollivier 1998
	Shifts from K-selected strategists to r-selected strategists following disturbance or in response to pollution	shifts from fragmentation-sensitive to fragmentation-tolerant bird species in relation to disturbed riparian habitats	Croonquist and Brooks 1993; Allen and O'Connor 2000; Bryce et al. 2002
		higher proportion of r-selected species in a flow regulated river as compared to a natural flow regime river	Nilsson et al. 1991
		shift to r-selected, generalist damselfly species along a gradient of increasing pollution	Solimini et al. 1997
		water-level fluctuation in a mesocosm resulted in increased proportion of r-strategist species	Troelstrup and Hengenrader 1990
		high pollutional stress correlated with increase in r-selected strategists in the same river 21 years apart	Richardson et al. 2000
	Regional and national species attribute lists and taxonomic tolerance values	compendium of pollution tolerance, habitat preferences, feeding guilds for fish species of the northeastern U.S.	Halliwell et al. 1998
		compendium of pollution tolerance, habitat preferences, feeding guilds for fish species of the Pacific northwest, U.S.	Zaroban et al. 1999
		organic pollution tolerance ranks for Wisconsin stream insect taxa	Hilsenhoff 1987
		compendium of pollution tolerance, habitat preferences, feeding guilds of North American fish and aquatic macroinvertebrate taxa	Barbour et al. 1999

TABLE 2-2. Evidence in support of the depicted changes in ecological attributes in the BCG.

BCG Attribute	Response	Case-specific documentation	Reference
VI	Detrimental effects of non-native taxa	loss of 150-200 endemic species in Lake Victoria following intentional introduction of Nile perch (*Lates niloticus*) and Nile tilapia (*Oreochromis niloticus*)	Witte et al. 1992
		dominance of many lowland rivers in the western USA by non-native fishes and invertebrates	Moyle 1986, Karr et al 1986, Miller et al. 1989
		food web disruption and loss of native mussels from zebra mussel invasion	Whittier et al. 1995
		detrimental changes in non-native taxa in TVA rivers where Corbicula is present	Kerans and Karr 1994
		loss of small, soft-finned fish species from Northeast USA lakes following predator introductions	Whittier and Kincaid 1999
		mid-twentieth century collapse of native salmonid fisheries following colonization of the Laurentian Great Lakes by sea lamprey (*Petromyzon marinus*) and alewife (*Alosa pseudoharengus*)	Smith 1972
VII	Changes in organism condition or increase in anomalies in response to pollution gradients	increased fish anomalies in the vicinity of toxic outfalls	Hughes and Gammon 1987, Yoder and Rankin 1995b
		altered blood chemistry and mortality in fish associated with wetlands that received oil sands effluent	Bendellyoung et al. 2000
		changes in growth, organism condition, fecundity, and feeding strategies for creek chub (*Semotilus atromaculatus*) across a variety of pressure gradients (urbanization, agriculture, temperature)	Fitzgerald et al. 1999
		the presence of tumors, deformities, lesions, etc. in the fish from highly disturbed streams	Karr et al. 1986, Yoder and DeShon 2003
VIII	ecosystem-level disruptions of functional integrity	extinction and succession of littoral lake invertebrate species secondary to lake acidification; initially detected by temporal changes in taxonomic and density measures but followed by top-down and bottom up effects at all trophic levels, caused by reduced nutrient cycling. A trophic cascade ultimately involved loss of fish and increased biomass of primary producers.	Appelberg et al. 1993
		simplification of global coastal ocean ecosystems to microbial domination due to combined effects of historical and current overfishing and pollution	Jackson et al. 2001
IX	influence of spatial and temporal scale of pressures on biological effects and recovery potential	large-scale, multi-state status and trends assessments of Pacific salmon influenced the listing of the species under the Endangered Species Act	Nehlsen et al. 1991
		environmental factors operating at different temporal and spatial scales influence the production and survivorship of juvenile Atlantic salmon	Poff and Huryn 1998
		past land use activity has long-term effects on aquatic bio-diversity	Harding et al. 1998
		assessments of stream fish and benthic macroinvertebrate assemblages at state and regional scales reveal serious alterations in indicators of biological integrity	U.S. EPA 2000a
		Ocean-wide ecological extinction of large predators from historical and current overfishing	Myers and Worm 2003
X	ecosystem connectance	replacement of 4 native freshwater fish species by 37 marine species in the lower Rio Grande following flow diversions that caused the lower river to cease flowing and become tidal salt water	Contreras-Balderas et al. 2002
		decreased fish species and guilds with decreased riverine connectivity with floodplain water bodies	Aarts et al. 2004
		5 federally listed headwater fish species have had their ranges restricted and isolated by mainstem impoundments, increasing their susceptibility to local physical and chemical habitat degradation	Freeman et al. 2005

TABLE 2-2. Evidence in support of the depicted changes in ecological attributes in the BCG.

BCG Attribute	Response	Case-specific documentation	Reference
		alteration of natural flow regimes result in changes in biological assemblage structure	Poff et al. 1997, Bunn and Arthington 2002
		extirpation of Pacific Northwest salmon following construction of impassable dams	Frissell 1993
		extirpation of Colorado River fishes following dam construction	Holden and Stalnaker 1975

Attribute VI: Non-native or intentionally introduced taxa.
With respect to a particular ecosystem, any species that is not found in that ecosystem. Species introduced or spread from one region of the U.S. to another outside their normal range are non-native or non-indigenous, as are species introduced from other countries.

This attribute represents both an effect of human activities and a stressor in the form of biological pollution. Although some intentionally introduced species are valued by large segments of society (e.g., gamefish), these species may be just as disruptive to native species as undesirable opportunistic invaders (e.g., zebra mussels). Many rivers in the U.S. are now dominated by non-native fishes and invertebrates (Moyle 1986), and introductions of alien species are the second most important factor contributing to fish extinctions in North America (Miller et al. 1989). The BCG identifies maintenance of native taxa as an essential characteristic of Tier 1 and 2 conditions. The model only allows for the occurrence of non-native taxa in these tiers if those taxa do not displace native taxa and do not have a detrimental effect on native structure and function. Tiers 3 and 4 depict increasing occurrence of non-native taxa. Extensive replacement of native taxa by tolerant or invasive, non-native taxa can occur in Tiers 5 and 6.

Organism Condition and System Performance: Attributes VII and VIII

Attribute VII: Organism condition.
Organism condition is an element of ecosystem function, expressed at the level of anatomical or physiological characteristics of individual organisms.

Organism condition includes direct and indirect indicators such as fecundity, morbidity, mortality, growth rates, and anomalies such as lesions, tumors, and deformities and for purposes of the BCG, primarily applies to fish and amphibians. Some of these indicators are readily observed in the field and laboratory, whereas the assessment of others requires specialized expertise and much greater effort. The most common approach for State and Tribal programs is to forego complex and demanding direct measures of organism condition (e.g., fecundity, morbidity, mortality, growth rates) in favor of indirect or surrogate measures (e.g., % of organisms with anomalies, age or size class distributions) (Simon (ed.) 2003). Organism anomalies in the BCG vary from naturally occurring incidence in Tiers 1 and 2 to higher than expected incidence in Tiers 3 and 4. In Tiers 5 and 6, biomass is reduced, the age structure of populations indicates premature mortality or unsuccessful reproduction, and the incidence of serious anomalies is high.

Attribute VIII: Ecosystem function.
"Function" refers to any processes required for normal performance of a biological system. The term may be applied to any level of biological organization. Immigration and emigration are functional processes at the population level. Examples of ecosystem functional processes are primary and secondary production, respiration, nutrient cycling, and decomposition.

The "functional integrity" of an ecosystem refers to the aggregate performance of dynamic interactions among an ecosystem's biological parts (Cairns 1977). The term "ecosystem function" includes measures of both the interactions among taxa (food web dynamics) and energy and nutrient processing rates (energy and nutrient dynamics). These attributes are included in the BCG because ecologists universally recognize their fundamental importance. At this time, the level of effort required to directly assess ecosystem function is beyond the means of most State and Tribal monitoring programs. Instead, most programs rely on taxonomic and structural indicators to make inferences about functional status (Karr et al. 1986). For example, shifts in the primary source of food may cause changes in trophic guild indexes or indicator species. Although direct measures of ecosystem function are currently difficult or time consuming, they may become practical in the future (Gessner and Chauvet 2002).

Attribute VIII also includes aspects of individual, population, and community condition. Altered interactions between individual organisms and their abiotic and biotic environments may generate changes in growth rates, reproductive success, movement, or mortality. These altered interactions are ultimately expressed at ecosystem-levels of organization (e.g., shifts from heterotrophy to autotrophy, onset of eutrophic conditions) and as changes in ecosystem process rates (e.g., photosynthesis, respiration, production, decomposition). Maine's example scenario (Table 2-3, located at the end of this chapter) describes a progression of functional changes. It depicts a naturally oligotrophic and heterotrophic system with P/R <1 in Tiers 1 and 2. Tiers 3 and 4 depict functional changes commonly associated with the effects of increased temperature and nutrient enrichment (P/R > 1, diurnal sags in dissolved oxygen, changes in taxonomic composition and relative abundance, increased algal biomass). Tier 5 depicts an autotrophic system impacted by excessive algal biomass.

Scale-dependent Factors: Attributes IX and X

Attribute IX: Spatial and temporal extent of stressor effects.
The spatial and temporal extent of stressor effects includes the near-field to far-field range of observable effects of the stressor. Patchy islands or periods of unsuitable conditions, within a generally intact system, give way to patchy islands or periods of suitable conditions, within a substantially degraded system.

Attribute X: Ecosystem connectance.
Access or linkage (in space/time) to materials, locations, and conditions required for maintenance of interacting populations of aquatic life; the opposite of fragmentation; necessary for metapopulation maintenance and natural flows of energy and nutrients across ecosystem boundaries.

***Scale-dependent factors* (ATTRIBUTES IX AND X).**
These attributes relate to interactions between the physical environment in all its aspects (spatial, temporal, structural, chemical, etc.), and the biota. Attributes IX and X are interpreted at different spatial and even temporal scales than the rest of the attributes, i.e., the reach, or sampled community perspective has been expanded to consider alterations occurring within entire catchments, basins, and regions, or within seasonal and annual cycles. These attributes were included in the BCG because the extent of ecosystem alteration has important environmental implications in terms of an individual waterbody's vulnerability to further effects from stressors as well as potential for mitigation. For example, ecosystem connectivity is fundamental to the successful recruitment and maintenance of organisms into any environment. A single impacted stream reach in an otherwise intact watershed has far more

restoration potential than a similar site in a basin that has undergone extensive land-scape alteration (Table 2-2). Tiers 1 and 2 depict a naturally connected or isolated system in which a natural disturbance regime, e.g. natural variability, is maintained. Detrimental effects in Tiers 3 and 4 are limited to the reach or seasonal scale. The two lowest tiers depict a system with detrimental effects extending to the catchment scale and affecting multiple seasons. A few "islands" of adequate physical/chemical conditions may serve as refugia in Tier 5, but extensive loss of connectance and refugia occur in Tier 6.

2.1.2 The BCG Tiers

Although the BCG is continuous in concept, it has been divided into six tiers to provide as much discrimination of different levels of condition as workgroup members deemed discernable, given current assessment methods and robust monitoring information (Figure 2-1). Defining the tiers between 3 and 5 was a challenge to the workgroup and entailed considerable discussion. The workgroup ultimately agreed some States and Tribes may only be capable of discriminating 3-4 tiers, while others might be capable of discerning 6 tiers based on characteristics of their database and monitoring program. However the workgroup agreed that the important role of the BCG model is to be a starting point for a State or Tribe to think about how to use information to better define their designated aquatic life uses and to communicate more clearly about biological condition. There is no expectation that States and Tribes establish six tiers of use classes. The ultimate number of the tiers is a State or Tribal determination.

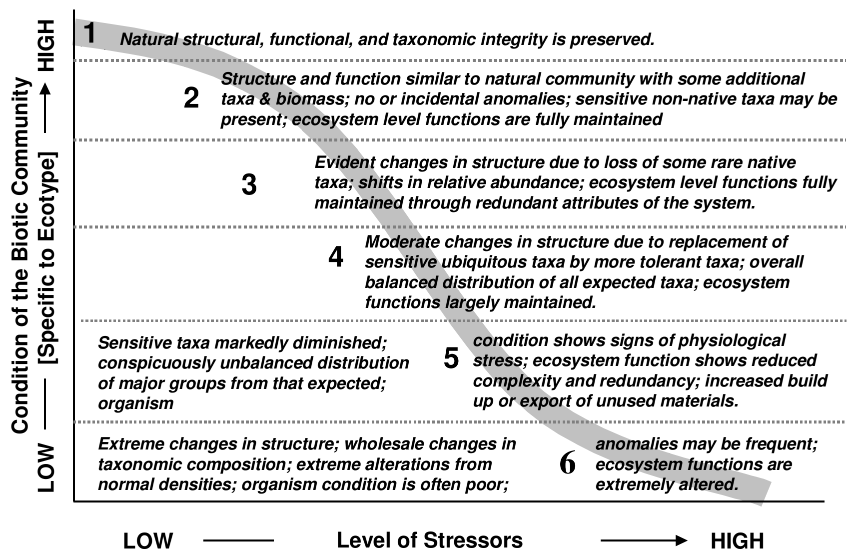


FIGURE 2-1. Conceptual model of the Biological Condition Gradient.

Tier 1: Natural or native condition.
Native structural, functional, and taxonomic integrity is preserved; ecosystem function is preserved within the range of natural variability.

Tier 1 represents biological conditions as they existed (or still exist) in the absence of measurable effects of stressors. The Tier 1 biological assemblages that occur in a given biogeophysical setting are the result of adaptive evolutionary processes and biogeography that selects in favor of survival of the observed species. For this reason, the expected Tier 1 assemblage of a stream from the arid southwest will be very different from that of a stream in the northern temperate forest. The maintenance of native species populations and the expected natural diversity of species are essential for Tiers 1 and 2. Non-native taxa (Attribute VI) may be present in Tier 1 if they cause no displacement of native taxa, although the practical uncertainties of this provision are acknowledged (discussed in Section 2.2).

Attributes I and II (e.g., historically documented and sensitive taxa) can be used to help assess the status of native taxa and could be a surrogate measure to identify threatened or endangered species when classifying a site or assessing its condition.

Tier 2: Minimal changes in structure of the biotic community and minimal changes in ecosystem function.
Virtually all native taxa are maintained with some changes in biomass and/or abundance; ecosystem functions are fully maintained within the range of natural variability.

Tier 2 represents the earliest changes in densities, species composition, and biomass that occur as a result of slight elevation in stressors (such as increased temperature regime or nutrient enrichment). There may be some reduction of a small fraction of highly sensitive or specialized taxa (Attribute II) or loss of some endemic or rare taxa as a result. Tier 2 can be characterized as the first change in condition from natural and it is most often manifested in nutrient enriched waters as slightly *increased* richness and density of sensitive ubiquitous taxa and taxa of intermediate tolerance (Attributes III and IV). These early response signals have been observed in many State programs as illustrated in Figure 2-2, showing slight to moderate increases in conductivity in Maine streams.

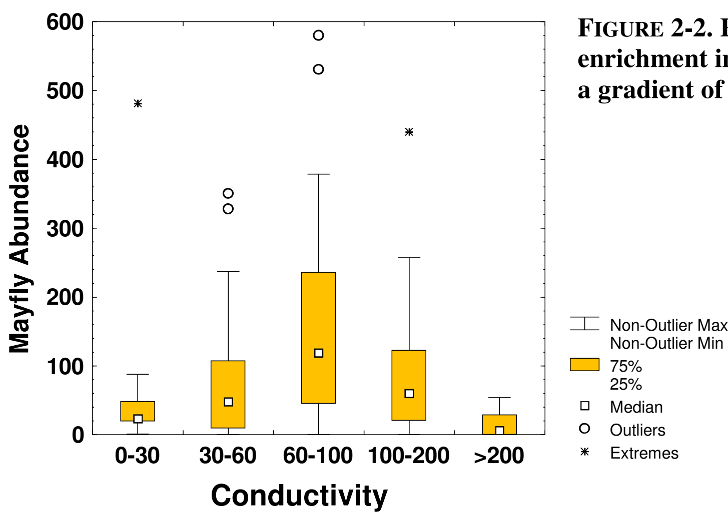


FIGURE 2-2. Response of mayfly density to enrichment in Maine streams as indicated by a gradient of increasing conductivity.

Tier 3: Evident changes in structure of the biotic community and minimal changes in ecosystem function.
Evident changes in structure due to loss of some rare native taxa; shifts in relative abundance of taxa but sensitive-ubiquitous taxa are common and abundant; ecosystem functions are fully maintained through redundant attributes of the system.

Tier 3 represents readily observable changes that, for example, can occur in response to organic enrichment or increased temperature. The "evident" change in structure for Tier 3 is interpreted to be perceptible and detectable decreases in sensitive-rare or highly sensitive taxa (Attribute II) and increases in sensitive-ubiquitous taxa or opportunist organisms (Attributes III and IV). Attribute IV taxa (intermediate tolerants) may increase in abundance as an opportunistic response to nutrient inputs.

Tier 4: Moderate changes in structure of the biotic community with minimal changes in ecosystem function.
Moderate changes in structure due to replacement of some sensitive-ubiquitous taxa by more tolerant taxa, but reproducing populations of some sensitive taxa are maintained; overall balanced distribution of all expected major groups; ecosystem functions largely maintained through redundant attributes.

Moderate changes of structure occur as stressor effects increase in Tier 4. A substantial reduction of the two sensitive attribute groups (II and III) and replacement by more tolerant taxa (Attributes IV and V) may be observed. A key consideration is that some Attribute III sensitive taxa are maintained at a reduced level but are still an important functional part of the system (function maintained).

Tier 5: Major changes in structure of the biotic community and moderate changes in ecosystem function.
Sensitive taxa are markedly diminished; conspicuously unbalanced distribution of major groups from those expected; organism condition shows signs of physiological stress; ecosystem function shows reduced complexity and redundancy; increased build-up or export of unused materials.

Changes in ecosystem function (as indicated by marked changes in food-web structure and guilds) are critical in distinguishing between Tiers 4 and 5. This could include the loss of functionally important sensitive taxa and keystone taxa (Attribute I, II and III taxa) such that they are no longer important players in the system, though a few individuals may be present. Keystone taxa control species composition and trophic interactions, and are often, but not always, top predators. As an example, removal of keystone taxa by overfishing has greatly altered the structure and function of many coastal ocean ecosystems (Jackson et al. 2001). Additionally, tolerant non-native taxa (Attribute VI) may dominate some assemblages and changes in organism condition (Attribute VII) may include significantly increased mortality, depressed fecundity, and/or increased frequency of lesions, tumors and deformities.

Tier 6: Severe changes in structure of the biotic community and major loss of ecosystem function.
Extreme changes in structure; wholesale changes in taxonomic composition; extreme alterations from normal densities and distributions; organism condition is often poor; ecosystem functions are severely altered.

Tier 6 systems are taxonomically depauperate (low diversity and/or reduced number of organisms) compared to the other tiers. For example, extremely high or low densities of organisms caused by excessive organic enrichment or severe toxicity may characterize Tier 6 systems.

2.2 How the BCG was developed, tested, and evaluated

In developing the BCG, the workgroup believed it was important that the model be grounded in sound theory, easy to apply, and meet the needs of practitioners around the country.

The BCG model was developed and tested by the TALU Workgroup. Based on recommendations from the full workgroup, a steering committee created a matrix that summarized biologists' experience and knowledge about how biological attributes change in response to stress in aquatic ecosystems (Table 2-1). In developing the BCG, the workgroup believed it was important that the model be grounded in sound theory as well as actual empirical observations, easy to apply, and meet the needs of users around the country. In building the model, the workgroup followed an iterative, inductive approach, similar to means-end analysis (Martinez 1998). The model was tested by determining how consistently workgroup members assigned samples of macroinvertebrates or fish to the six tiers, the results of which support the contention that the BCG represents aspects of biological condition common to all existing assessment methods.

The workgroup began by testing whether biologists from different parts of the country would draw similar conclusions regarding the condition of a waterbody using simple lists of organisms and their counts. This approach was based on Maine's experience, in which expert biologists independently assigned samples of macroinvertebrates to *a priori* defined classes of biological condition defined by differences in assemblage attributes (Davies et al. 1995). Decision instructions were provided to biologists in the form of a matrix, which outlined expected trajectories of quantifiable aspects of invertebrates *(See Case Example 3-3 in the next chapter)*. These corresponded with biological expectations for four water quality classes (A, B, C and Non-Attainment; *See Appendix A, Tables A-1 and A-2*). The high level of majority and unanimous agreement (98% and 64% respectively) among experts in placing samples into the different classes allowed Maine to develop a predictive statistical model that is now used to assess the biological condition of new sites (Courtemanch 1995) (*See Case Example 3-3)*.

To provide a functional framework for practitioners, the TALU Workgroup described how each of the ten attributes varies across six tiers of biological alteration (Table 2-1). The general model was then described in terms of the biota of a specific region (Maine). Based on 20 years of biomonitoring data, the Maine example describes how the relative densities of specific taxa with varying sensitivities to stressors change across the BCG tiers (Table 2-3, located at the end of this chapter).

To test the general applicability of the BCG to sampling data taken from real ecosystems, the workgroup evaluated how consistently individual biologists classified samples of aquatic biota based on the attributes incorporated into the BCG. Governmental and research biologists from 23 States and one Tribe participated in the data exercise. The full workgroup was divided into breakout groups according to regional (Northeast, South-Central, Northwest, Arid Southwest/Great Plains) or assemblage (fish, invertebrates) expertise. Samples were selected from invertebrate and fish data sets to span as many of the BCG tiers as possible. The invertebrate samples and fish samples used in the tests were collected from six different regions within the U.S. (Northeast, Mid-Atlantic, Southeast, Northwest, Southwest, Central) and included only basic descriptors of stream physical characteristics (substrate, velocity, width, depth, etc.), taxonomic names, densities, and in some cases, metric values. These data represent the basic core elements common to nearly all biological monitoring programs. Participants were asked to place each sample into one of the six condition tiers, though they were cautioned not to apply a simple relative quality ranking since all six tiers did not necessarily occur within the data sets. Biologists relied primarily on differences in relative abundances and sensitivities of taxa (i.e., Attributes I-VI) to make tier assignments because information needed to evaluate the status of the other Attributes was not available. Percent concurrence among the individuals was calculated to assess the level of agreement among biologists when applying the BCG to raw data. Perfect concurrence was set to equal the product of the number of raters by the number of streams. *Case Examples 3-2, 3-3, and 3-7, at the end of Chapter 3, outline how Maine and New Jersey biologists described tiers and assigned sites.*

In the first stage of the data exercise, between-biologist differences were evaluated by asking workgroup participants to rate a single data set of 6-8 samples. The breakout groups were then asked to classify samples from larger and more variable datasets. The groups were also instructed to summarize their interpretations and to identify biological responses to changes in conditions not captured by the BCG. Finally, the groups identified which tiers corresponded to how they currently assess biological integrity and the CWA interim goal for protection and propagation of aquatic life.

Workgroup members placed 82% of the benthic macroinvertebrate samples and 74% of the fish samples into the same BCG tiers. The range of variation among individuals was within one tier's distance in either direction. Tiers were revised following full workgroup discussion so that transitions were more distinct. Each of the breakout groups independently reported that the ecological characteristics approximately described by Tiers 4 and above were compatible with how they currently assess the CWA's interim goal for protection and propagation of aquatic life. These groups also identified the characteristics described by Tiers 1 and 2 as indicative of biological integrity.

Workgroup members reported that key concepts were important with respect to classifying samples into tiers and identifying the boundaries in between. For Tiers 1 and 2, biologists identified the maintenance of native species populations as essential to their understanding of biological integrity. Although many participants noted that criteria for distinguishing differences between tiers in Attribute VIII (ecosystem function) were poorly defined, most nevertheless identified ecosystem function changes (as indicated by marked changes in food-web structure and guilds) as critical in distinguishing between Tiers 4 and 5.

Discussion following the BCG exercise revealed that participants readily agreed on some of the condition attributes, but not others. For example, participants indicated they mostly used Attributes I-V (taxonomic composition and tolerance), Attribute VI (non-native taxa, for Tiers 2-6 only) and Attribute VII (organism condition) to evaluate biological conditions. In contrast, because Attributes VIII - X (ecosystem function and scale-dependent features) are rarely directly assessed by biologists, the evaluation of these attributes was accompanied by relatively high uncertainty. Even so, workgroup members strongly advocated retaining these attributes in the BCG because of the importance of this information in making restoration decisions.

The presence of non-native taxa in Tier 1 was also the subject of considerable discussion. Knowledge of the extensive occurrence of some non-native taxa in otherwise near-pristine systems conflicted with the desire by many to maintain a conceptually pure and natural tier. Further discussion resulted in agreement that the presence of non-native taxa in Tier 1 is permissible only if they cause no displacement of native taxa, although the practical uncertainties of this provision were acknowledged. The resulting tier descriptions, which allow for non-native species in the highest tiers as long as there is no detrimental effect on the native populations, has practical management implications. For example, introduced European brown trout (*Salmo trutta*) have replaced native brook trout (*Salvelinus fontinalis*) in many eastern U.S. streams. In some catchments, brook trout only persist in stream reaches above waterfalls that are barriers to brown trout. The downstream reaches are nearly pristine except for the presence of brown trout (D. Lenat, North Carolina Department of Natural Resources, personal communication). In these places, if society decided to remove the introduced brown trout and if stream habitat is preserved throughout the catchment, brook trout can potentially repopulate downstream reaches. In the use designation process, recognizing that the entire catchment has the *potential* to attain Tier 1 conditions will inform the public that a very high quality resource exists.

Critical gaps in knowledge were uncovered during the development of the BCG. For example, the workgroup identified the need for regional evaluations of species tolerance to stressors associated with pressure. Tolerance information presented in the current version of the BCG tends to be based on generalized taxa responses to a non-specific stressor gradient. At this time, tolerance information is not available for most taxa and for many common stressors (temperature, nutrients, sediments). In some cases, tolerance values are based on data collected in other geographic regions or for other purposes (e.g., van Dam's European diatom tolerances are used for North American taxa) (van Dam et al. 1994). Improved tolerance value information is needed to refine the BCG and improve its precision.

Additionally, taxa that are considered tolerant to stressors in one region of the country may not be similarly classified in another region. For example, long-lived taxa have generally been characterized as sensitive to increasing pressure and tend to be replaced by short-lived taxa in stressed systems. As such, the presence of long-lived taxa in a waterbody has been used to indicate high quality conditions, whereas the predominance of short-lived taxa indicates degradation. However, in small streams in the arid western U.S., extreme changes in hydrology define the natural regime for some systems and an opposite trend has been observed: short-lived taxa can dominate the biological community in natural settings. In these systems, a shift to long-lived taxa may be an indicator of altered, less variable flow regimes.

2.3 The relationship between the BCG and designated uses

The BCG is a model that provides a rational and consistent way to identify and communicate waterbody condition. It can thus be used to establish appropriate ALUs in State water quality standards and to assess attainment. The ecological condition to support an ALU for a specific waterbody can be described in terms of the BCG tiers and can be related to specific use categories such as fishery-based uses. For example, the ecological condition needed to support salmon spawning is an exceptional, high-quality natural stream and will likely be either a Tier 1 or 2 on the BCG. The ecological attributes that characterize the BCG tiers can be measured with methods used by each State, and these condition assessments can be directly linked to a State's ALUs.

Maine and Ohio are examples of States that have adopted uses based on a biological condition gradient into water quality standards (Courtemanch et al. 1989, Yoder and Rankin 1995a). Both of these States have incorporated multiple tiers of resource quality in their water quality standards (State of Maine 1985, 2003; Davies et al. 1995; State of Ohio 2003). As discussed above, the tiers in these States' TALUs describe aquatic-life management goals and attainment criteria for different waterbody types. For example, in Maine a waterbody is assigned to one of four management tiers by considering both its existing biological condition and its highest attainable condition as determined by a public and legislative process. These four tiers of biological quality in Maine's water quality standards are based on Odum's subsidy stress gradient (Odum et al. 1979, Odum 1985) (S*ee Appendix A, Figure A-2a and Table A-1*). Attainment of standards is assessed by determining to which tier a sample of macroinvertebrates is most similar (Courtemanch et al. 1989). Site-specific taxonomic composition data and other metrics are used in a discriminant model to identify the class of a particular waterbody (*See Case Examples 3-3 and 3-6 in Chapter 3*). Maine has found multiple tiers to be useful in 5 ways:

1) identifying and preserving the highest quality resources,
2) depicting existing conditions more accurately,
3) setting realistic and attainable management goals,
4) preserving incremental improvements, and
5) determining appropriate management action when conditions decline.

Over the past thirty years, States have independently developed technical approaches to assess condition and set ALUs specific to the biology of the State and its regulatory and political settings (U.S. EPA 2002a). Although these different approaches have fostered innovative technical approaches, they have also complicated the development of a nationally consistent approach to interpreting the condition of aquatic resources. Assessment results are often difficult to compare when quantitative outcomes (i.e., index or indicator values) represent different qualitative conditions. Additionally, without a common interpretative framework, use of different methods can hinder collaboration among natural resource agencies that have complementary missions. A consistent approach to interpreting biological condition will allow scientists and the public to more effectively evaluate the current and potential conditions of specific waters and watersheds and use that information to set appropriate ALUs.

The BCG can help promote consistent interpretation of scientific data by applying a common framework to diverse conditions and different assessment methods at national, regional, state, or watershed levels. By providing a means for managers and the public to identify outstanding resources, recognize incremental improvements, more appropriately allocate resources and prioritize management actions, aquatic and natural resource agencies will be able to coordinate and target resources more effectively.

2.4 Key points from Chapter 2

1. **The biological condition gradient is a descriptive model predicting biological response to increasing levels of stressors**. The biological gradient can be thought of a field-based dose-response curve where dose (x-axis) is level of stressors and response (y-axis) is biological condition.

2. **The purpose of the Biological Condition Gradient is to provide an ecologically-based model about biological condition and to promote clearer understanding of current conditions relative to natural conditions**. This should result in more meaningful engagement of the public in the designation of aquatic life uses in State and Tribal water quality standards programs.

3. **The model must be validated with data**. The BCG model does not reduce the necessity of developing robust methods for the quantitative and statistical validation of biological conditions. The list of attributes is intended to organize how we interpret biological information concerning a given aquatic community response to increasing levels of stressors. The approach should be thought of as seeking to identify a "best fit" tier, which consists of weighing the importance and signal-strength of the different attributes as they pertain to a specific waterbody or as used to describe a designated use class.

4. **The conceptual framework is not defined by any one method**. As presented in Chapter 3, the attributes have a quantifiable aspect that can potentially be assessed and validated in many different ways. The BCG has been designed to be independent of different assessment methodologies (i.e. Rapid Biological Assessment, Index of Biological Integrity; RIVPACS, multivariate analyses, etc.). The intent is for the ecological premises that support the model to reflect the same basis that underlies all successful methods used to quantify biological response to increasing levels of stressors.

5. **The number of useful tiers is flexible**. The purpose of the number of tiers is to provide a highly resolved biological condition gradient. There is no expectation that State or Tribal programs adopt six tiers, or categories, of designated uses. While step-wise progress toward refinement of designated aquatic life uses in State and Tribal water quality standards programs is desired over the long term, the ultimate number and type of tiers of uses is a State or Tribal determination.

6. **The BCG was designed to facilitate communication of the current biological condition of a waterbody compared to natural conditions**. For example, the BCG is grounded in natural conditions, which can help users and the public understand that current conditions do not necessarily represent natural conditions. In areas where natural or near-natural conditions exist, people are generally familiar with what is natural and what is altered. But in extensively altered regions practitioners and the public alike tend to accept the "best of what is left" as the potential for a system. In such places, it is difficult to visualize the natural conditions that were once present and designated uses may end up based on a diminished perspective. Natural conditions may not be achievable in many places, but an improved understanding of the changes that have occurred will result in a more scientifically defensible evaluation of current conditions and what can potentially be restored.

The next chapter provides information on how to adapt the national BCG model to reflect the specific ecology and stressor gradient characteristics of a particular state or region, and introduces some ways to quantify a biological condition gradient with monitoring data.

TABLE 2-3. Biological Condition Gradient: Maine example scenario for a cold-water stream catchment. [1]

Resource Condition "Tiers"	**Biological Condition Characteristics (Effects)**
1 **Natural or native condition** *Native structural, functional and taxonomic integrity is preserved; ecosystem function is preserved within the range of natural variability*	**I *Historically documented, sensitive, long-lived, or regionally endemic taxa*** ➔ Long-lived native species of fish-host specialist or long-term brooder mussels such as Brook floater- *Alasmodonta varicosa;* Triangle floater- *Alasmodonta undulata;* Yellow lampmussel- *Lampsilis cariosa* are present in naturally occurring densities ➔ **Fishes**: Brook stickleback, Swamp darter **II *Sensitive- rare taxa*** ➔ The proportion of total richness represented by rare, specialist and vulnerable taxa is high, for example, without limitation, the following taxa are representative: **Plecoptera**: Capniidae, *Taeniopteryx, Isoperla, Perlesta, Pteronarcys, Leuctra*; **Ephemeroptera**: *Cinygmula, Rhithrogena, Epeorus, Serratella, Leucrocuta*; **Trichoptera**: *Glossosoma; Psilotreta; Brachycentrus*; **Diptera**: *Stempellina, Hexatoma, Probezzia*; **Coleoptera**: *Promoresia;* **Fishes***:* Slimy sculpin, Longnose sucker; Longnose dace **III *Sensitive- ubiquitous taxa*** ➔ Densities of Sensitive-ubiquitous taxa are as naturally occur. The following taxa are representative of this group for Maine: **Plecoptera**: *Acroneuria*; **Ephemeroptera**: *Stenonema, Baetis, Ephemerella, Pseudocloeon*; **Fishes**: Brook trout, Burbot, Lake chub **IV *Taxa of intermediate tolerance*** ➔ Densities of *intermediate tolerance taxa* are as naturally occur. The following taxa are representative of this category: **Trichoptera**: Hydropsychidae, Chimarra, *Neureclipsis*, *Polycentropus;* **Diptera**: *Tvetenia, Microtendipes, Rheocricotopus, Simulium;* **Fishes**: Common shiner, Fallfish **V *Tolerant taxa*** ➔ Occurrence and densities of Tolerant taxa are as naturally occur. The following taxa are representative of this category: **Diptera**: *Dicrotendipes, Tribelos, Chironomus, Parachironomus*; **Non-Insects**: Caecidotea, Isopoda, *Physa*, *Helobdella;* **Fishes***:* White sucker, Blacknose dace, Creek chub **VI *Non native or intentionally introduced taxa*** ➔ Non native taxa such as Brown trout, Rainbow trout, Yellow perch, are absent or, if they occur, their presence does not displace native biota or alter native structure and function **VII *Physiological condition of long-lived organisms*** ➔ Anomalies are absent or rare; any that occur are consistent with naturally occurring incidence and characteristics **VIII *Ecosystem Function*** ➔ Rates and characteristics of *life history (e.g., reproduction, immigration, mortality, etc.)*, and materials exchange processes *(e.g., production, respiration, nutrient exchange, decomposition, etc.)* are comparable to that of "natural" systems ➔ The system is predominantly heterotrophic, sustained by leaf litter inputs from intact riparian areas, with low algal biomass; P/R<1 (Photosynthesis: Respiration ratio) **IX *Spatial and temporal extent of detrimental effects*** ➔ Not applicable- disturbance is limited to natural events such as storms, droughts, fire, earth-flows. A natural flow regime is maintained. **X *Ecosystem connectance*** ➔ Reach is highly connected with groundwater, its floodplain, and riparian zone, and other reaches in the basin, at least annually. Allows for access to habitats and maintenance of seasonal cycles that are necessary for life history requirements, colonization sources and *refugia* for extreme events.

[1] This scenario presents Maine biologists' summary of the ecological characteristics of the six tiers in the Biological Condition Gradient model as observed in Maine (see Appendix A, Sections II and III). It is based on analysis of genus/species level benthic macroinvertebrate data (400 samples from rivers and streams spanning conditions from near-natural to severely altered) (Davies et al. 1999).

2

Minimal changes in structure of the biotic community and minimal changes in ecosystem function

Virtually all native taxa are maintained with some changes in biomass and/or abundance; ecosystem functions are fully maintained within the range of natural variability

I *Historically documented, sensitive, long-lived, regionally endemic taxa*

➔ Some regionally endemic, long-lived species (e.g., some mussel species such as the Dwarf wedgemussel- *Alasmidonta heterodon*, and/or fish species, such as the Brook stickleback are absent due extirpation from Maine prior to the enactment of the CWA; some mussel species of Special Concern in Maine are present (e.g., Brook floater- *Alasmodonta varicosa;* Triangle floater- *Alasmodonta undulata;* Yellow lampmussel- *Lampsilis cariosa*)

II *Sensitive- rare taxa*

➔ Richness of rare and/or specialist invertebrate taxa is high though densities may be low (e.g., for Maine- **Plecoptera**: Capniidae, *Taeniopteryx, Isoperla, Agnetina, Perlesta, Pteronarcys, Leuctra*; **Ephemeroptera**: *Cinygmula, Rhithrogena, Epeorus, Serratella, Leucrocuta*; **Trichoptera**: *Glossosoma, Psilotreta, Brachycentrus*; **Diptera**: *Stempellina, Rheopelopia, Hexatoma, Probezzia*; **Coleoptera**: *Promoresia*). Densities of scrapers such as Glossosoma are increased

➔ Fish assemblage is predominantly native including such sensitive fish as Slimy sculpin, Longnose sucker, Longnose dace.

III *Sensitive- ubiquitous taxa*

➔ Superficial scraper-grazers and collector-gathers are favored due to slightly increased periphyton biomass on hard substrates, which results in higher relative abundance of these groups (e.g., **Ephemeroptera**: *Stenonema, Stenacron, Baetis, Ephemerella, Pseudocloeon*). Predatory stoneflies are common (e.g., *Acroneuria, Agnetina*). Populations of such native fish taxa as Brook trout, Lake chub, Burbot are common.

***IV* Taxa of intermediate tolerance**

➔ Increased biomass of diatom species that respond positively to increased nutrients and temperatures, but sensitive diatom species are maintained. Diatom richness is increased; filamentous forms are rare or as naturally occur

➔ May be slight increases in densities of macroinvertebrate taxa such as **Trichoptera**: Hydropsychidae, Philopotamidae, *Neureclipsis;* **Diptera**: *Rheotanytarsus, Microtendipes, Rheocricotopus, Simulium*

➔ Common shiner and Fallfish are in good condition

V *Tolerant taxa*

➔ May be slight increases in occurrence of tolerant taxa such as **Diptera**: *Polypedilum, Tvetenia,* **Non-Insects**: *Isopoda, Physa*; **Fishes**: White sucker; Creek chub, Blacknose dace

VI *Non-native or intentionally introduced* taxa

➔ Any intentionally introduced fish species (e.g., Brown trout- *Salmo trutta*, Rainbow trout- *Oncorhynchus mykiss)* occupy non-detrimental niche space

VII *Physiological condition of long-lived organisms*

➔ Any anomalies on fish are consistent with naturally occurring incidences and characteristics such as rare occurrence of gill or anchor parasites, blackspot, etc.

➔ Spawning areas of native fishes are evident during spawning season

VIII *Ecosystem Function*

➔ Rates and characteristics of *life history* (e.g., reproduction; immigration; mortality etc.), and materials exchange processes (e.g., production; respiration; nutrient exchange; decomposition e*tc.)* are unimpaired and not significantly different from the range of natural variability.

➔ The system is predominantly heterotrophic, sustained by leaf litter inputs from intact riparian areas; P/R/ is<1

IX *Spatial and temporal extent of detrimental effects*

➔ Extent is limited to small pockets or brief periods

X *Ecosystem connectance*

➔ Unimpaired access to habitats and maintenance of seasonal cycles that are necessary to fulfill *life history requirements*, and to provide colonization sources and *refugia* for extreme events.

3 **Evident changes in structure of the biotic community and minimal changes in ecosystem function** *Some changes in structure due to loss of some rare native taxa; shifts in relative abundance of taxa but <u>sensitive-ubiquitous</u> taxa are common and abundant; ecosystem functions are fully maintained through redundant attributes of the system*	**I *Historically documented, sensitive, long-lived, or regionally endemic* taxa** ➔ Brook floater- *Alasmodonta varicosa;* Triangle floater- *Alasmodonta undulata;* Yellow lampmussel- *Lampsilis cariosa;* are uncommon; Dwarf wedgemussel- *Alasmidonta heterodon* (and/ or a fish species) absent due to extirpation from Maine prior to CWA **II *Sensitive- rare taxa*** ➔ Some replacement of taxa having narrow or specialized environmental requirements, with functionally equivalent *sensitive-ubiquitous* taxa; coldwater obligate taxa are disadvantaged. Taxa such as **Plecoptera**: Capniidae, *Taeniopteryx, Isoperla, Perlesta, Pteronarcys, Leuctra*, *Agnetina*; **Ephemeroptera**: *Cinygmula, Rhithrogena, Epeorus, Serratella, Leucrocuta*; **Trichoptera**: *Glossosoma, Psilotreta, Brachycentrus*; **Diptera**: *Stempellina, Rheopelopia; Hexatoma, Probezzia*; **Coleoptera**: *Promoresia;* **Fishes**: Brook stickleback, Longnose sucker, Longnose dace are uncommonly encountered or absent **III *Sensitive- ubiquitous or generalist* taxa** ➔ Sensitive- ubiquitous or generalist taxa are common and abundant; taxa with broader temperature-tolerance range are favored (e.g., **Plecoptera**: *Acroneuria*; **Ephemeroptera**: *Stenonema, Baetis, Ephemerella, Pseudocloeon*) ➔ Overall mayfly taxonomic richness is reduced relative to the Tier 2 condition, with the preponderance of richness represented by sensitive- ubiquitous taxa; densities of remaining taxa are high and are sufficient to indicate healthy, reproducing populations ➔ Native Brook trout are significantly reduced due to the introduction of non-native Brown trout and the increased temperature regime **IV *Opportunist or facultative taxa of intermediate tolerance*** ➔ Filter-feeding blackflies (*Simulium*) and net-spinning caddisflies (e.g., *Hydropsyche, Cheumatopsyche, Polycentropus, Neureclipsis*) show increased densities in response to nutrient enrichment, but relative abundance of all expected major groups is well-distributed ➔ Increased temperature and increased available nutrients result in increased algal productivity causing an increase in the thickness of the diatom mat. This results in a "slimy" covering on hard substrates. ➔ Fish assemblage exhibits increased occurrence of Common shiner and Fallfish **V *Tolerant taxa*** ➔ Richness of **Diptera**: Chironomidae is increased; relative abundance of Diptera and Non-insects is somewhat increased but overall relative abundance is well-distributed among taxa from Groups III, IV and V, with the majority of taxa represented from Groups III and IV. Blacknose dace and white sucker are more common. **VI *Non-native or intentionally introduced* taxa** ➔ Brown trout have largely replaced native brook trout **VII** *Physiological condition of long-lived organisms* ➔ Incidence of *anomalies* such as gill parasites, anchor parasites, blackspot, etc., is low; serious anomalies such as tumors or deformities are essentially absent ➔ Environmental quality is sufficient to fully support reproduction of most long-lived species **VIII *Ecosystem Function*** ➔ Increased temperature and algal metabolism causes small diurnal sags in dissolved oxygen, compensated by adequate aeration from turbulence over riffle areas ➔ Algal biomass somewhat exceeds what can be utilized by resident grazers, resulting in evidence of die-back and slight downstream export of sloughed material. ➔ Patchy loss of high food quality riparian vegetation (e.g., oak; maple, beech) and elevated temperature, results in decreased growth and survival of some specialized shredder taxa (Pteronarcidae; Taeniopterygidae) with replacement by shredders capable of utilizing lower quality organic matter (Lepidostomatidae; Limnephilidae; Tipulidae). **IX *Spatial and temporal extent of detrimental effects*** ➔ Filamentous green algae occur in small patches within reaches; low dissolved oxygen levels occur only during the high temperature and low flow summer periods. ➔ Interstitial spaces, within the substrate of pools, are filled with fine sediment resulting in localized losses of interstitial habitats but riffle areas continue to provide adequate water flow and oxygen through interstitial habitats. **X *Ecosystem connectance*** ➔ Some downcutting has resulted in a patchy decrease in *connectance* of the stream from its floodplain except at unusually high flows. ➔ Thinning and patchy loss of riparian vegetation has altered the microclimate of the surrounding landscape causing a decrease in survival and reproductive success of adult mayflies and stoneflies.

4 **Moderate changes in structure of the biotic community and minimal changes in ecosystem function** *Moderate changes in structure due to replacement of some Sensitive-ubiquitous taxa by more tolerant taxa, but reproducing populations of some Sensitive taxa are maintained; overall balanced distribution of all expected major groups; ecosystem functions largely maintained through redundant attributes*	**I Historically documented, *sensitive, long-lived, regionally endemic* taxa** ➔ Healthy, reproducing populations of generalist mussel species are present (such as Eastern elliptio- *Eliptio complanata*; or Eastern lampmussel- *Lampsilis radiata radiata or* Eastern floater- *Pyganodon cataracta*) but Brook floater- *Alasmodonta varicosa;* Triangle floater- *Alasmodonta undulata;* Yellow lampmussel- *Lampsilis cariosa* are absent. **II *Sensitive- rare, specialist, vulnerable taxa with narrow environmental requirements*** ➔ Richness of specialist and vulnerable taxa is notably reduced; if present, densities are low (e.g., **Plecoptera**: Capniidae, *Taeniopteryx, Isoperla, Perlesta, Pteronarcys, Leuctra*; *Agnetina*; **Ephemeroptera**: *Cinygmula, Rhithrogena, Epeorus, Serratella, Leucrocuta*; **Trichoptera**: *Glossosoma; Psilotreta; Brachycentrus*; **Diptera**: *Stempellina, Rheopelopia; Hexatoma, Probezzia*; **Coleoptera**: *Promoresia,* **Fishes***:* Occurrence of Slimy sculpin, Longnose sucker and Longnose dace is reduced **III *Sensitive- ubiquitous or generalist taxa*** ➔ Densities of sensitive- ubiquitous scraper and gatherer insects (e.g., *Stenonema, Heptagenia, Baetis, Ephemerella, Pseudocloeon*) are sufficient to indicate that reproducing populations are present but relative abundance is reduced due to increased densities of opportunist invertebrate taxa (Group IV); ➔ Predatory stoneflies are reduced (e.g., *Acroneuria*) **IV *Opportunist or facultative taxa of intermediate tolerance*** ➔ Many substrate surfaces are covered by bryophytes and macro-algae responding to increased nutrients, resulting in displacement of lithophytic (stone-dwelling) micro-algae in favor of epiphytic (plant-dwelling) and filamentous forms (e.g., *Cladophora*). ➔ Increased loads of suspended particles favor collector-filterer invertebrates resulting in notably increased densities and relative abundance of filter-feeding caddisflies and chironomids (e.g., **Trichoptera**: Hydropsychidae, *Chimarra*, *Neureclipsis*, *Polycentropus;* **Diptera***: Tvetenia, Microtendipes, Rheocricotopus, Simulium;* **Fishes***:* Common shiner and Fallfish are common and abundant **V *Tolerant taxa*** ➔ There is an increase in the relative abundance of tolerant generalists (for example, *Polypedilum, Eukeifferiella, Cricoptopus*) and/or in numbers of non-insect scrapers and gatherers (e.g., *Physa, Sphaerium, Asellus, Hyalella)* but they do not exhibit significant dominance ➔ Overall relative abundance is well distributed among taxa from Groups III, IV and V, with the majority of the total abundance represented from Group IV. ➔ **Native fish such as White sucker, Blacknose dace, Creek chub are common.** **VI *Non-native or intentionally introduced* taxa** ➔ **Brook trout are absent or transient but such taxa as Smallmouth bass, Golden shiner and Yellow perch are common.** **VII** Physiological condition of long-lived organisms ➔ Incidence of anomalies such as blackspot and gill and anchor parasites is slightly higher than expected ➔ Occurrence of tumors, lesions and deformities is rare **VIII Ecosystem Function** ➔ Increased available nutrients increase algal productivity causing increased diatom, macro-algae and macrophyte biomass, and consequently lowering evening dissolved oxygen levels and increasing daytime oxygen levels. Invertebrate biomass is high but production has shifted to result in greater biomass of intermediate tolerance organisms than sensitive organisms. For example, filter-feeders utilizing suspended material shift from mayflies and sensitive mussels and caddisflies (e.g., *Isonychia, Elliptio, Brachycentrus*) to facultative types (e.g., Hydropsychidae, *Rheotanytarsus*, Sphaeriidae, *Musculium, Pisidium*); grazers of diatoms shift from sensitive mayflies and caddisflies (e.g., *Heptagenia, Leucrocuta*, Glossosomatidae) to facultative scrapers and collector gatherer organisms (e.g., *Baetis*, *Callibaetis*, Physidae, Leptoceridae). The suspended organic matter load somewhat exceeds what can be utilized by resident filterers resulting in increased levels of exported material. Sloughing of excess macro-algae and macrophyte biomass results in increased downstream export of course particulate organic matter. ➔ The system is becoming more autotrophic due to algal photosynthesis. The P/R ratio shows a slight increase. **IX *Spatial and temporal extent of detrimental effects*** ➔ Increased macrophyte and algal biomass extends downstream beyond the confluence with the next tributary; filamentous algae first appears in the stream as temperatures warm in late spring; pools and depositional areas are silt-filled; the interstitial spaces in the substrate of runs is becoming obstructed by sand and silt ➔ Early morning low dissolved oxygen levels occur occasionally during late spring and fall as well as during the mid summer **X *Ecosystem connectance*** ➔ Filling of interstitial spaces obstructs access to hyporheic zone for early instar stonefly nymphs, eliminating nursery areas and *refugia* for storm-events and low flows. Adult stoneflies from upstream reaches continue to oviposit but reproductive success is limited; stonefly nymphs continue to colonize by drift, with limited success. ➔ Poorly managed culverts on some tributaries impede fish passage and access to some spawning areas.

5

Major changes in *structure* of the biotic community and moderate changes in *ecosystem function*

Sensitive taxa are markedly diminished; conspicuously unbalanced distribution of major groups from that expected; organism condition shows signs of physiological stress; system function shows reduced complexity and redundancy; increased build-up or export of unused materials

I *Historically documented, sensitive, long-lived, or regionally endemic taxa*

- ➔ Mussel fauna, including commonly occurring, generalist taxa (e.g., Eastern lampmussel- *Lampsilis radiata radiata;* Eastern floater- *Pyganodon cataracta;* Eastern elliptio- *Elliptio complanata*) is markedly diminished due to poor water quality

II *Sensitive- rare taxa*

- ➔ Only the rare occurrence of individual representatives of specialist and vulnerable taxa with no evidence of successful reproduction

III *Sensitive- ubiquitous taxa*

- ➔ Either absent or present in very low numbers, indicating impaired recruitment and/or reproduction

IV *Opportunist or facultative taxa of intermediate tolerance*

- ➔ Filter-feeding invertebrates such as Hydropsychid caddisflies (e.g., *Cheumatopsyche*) and filter-feeding midges (e.g., *Rheotanytarsus, Microtendipes*) occur in very high numbers

V *Tolerant taxa*

- ➔ Frequent occurrence of tolerant collector-gatherers (e.g., Orthocladiini, *Micropsectra, Pseudochironomus, Dicrotendipes,* Isopoda- *Caecidotea;* Amphipoda- *Hyalella, Gammarus*);
- ➔ Relative abundance of non-insects often equal to or higher than relative abundance of insects
- ➔ Deposit-feeders such as Oligochaeta are increased
- ➔ Numbers of tolerant predators are increased (Hirudinea, *Thienemannimyia, Cryptochironomus*)
- ➔ Native fish species are essentially absent with the exception of tolerant taxa like White sucker, Blacknose dace and Creek chub

VI *Non-native or intentionally introduced taxa*

- ➔ Golden shiner, Smallmouth bass, and Yellow perch are common

VII Physiological condition of long-lived organisms

- ➔ Biomass of young of year age classes is low; overall fish biomass is reduced;
- ➔ Sex ratio of remaining fish does not equal 1
- ➔ Occurrence of parasitic infestations and disease is common
- ➔ Incidence of serious anomalies such as tumors and anatomical deformities is higher than expected

VIII *Ecosystem Function*

- ➔ High algal photosynthetic activity results in daytime dissolved oxygen supersaturation accompanied by nighttime dissolved oxygen levels less than 4 ppm. Extremely high algal biomass significantly alters the habitat structure of the substrate;
- ➔ The P/R ratio is significantly > 1; the system is predominantly autotrophic
- ➔ Loss of coarse particulate shredders and alteration of bacterial decomposer community contributes to build-up and/or export of unused organic matter;
- ➔ Mechanisms for nutrient spiraling are significantly simplified and less efficient resulting in increased export of nutrients from the system

IX Spatial and temporal extent of detrimental effects

- ➔ Substrate has become armored by increased sediment loading, altered flow regime and altered channel morphology resulting in compaction of interstitial space habitat, leaving only patches of well-scoured gravel substrate in high-gradient riffle areas;
- ➔ Armoring is resistant to spring scouring events, preventing annual spring sediment flushing and re-sorting of substrate;
- ➔ Near complete canopy removal results in all day insolation of stream and surrounding land surface causing abnormally elevated temperature regime in early spring and late fall. This causes unnaturally elevated seasonal temperature cues and results in failures of *life history requirements.*

X *Ecosystem connectance*

- ➔ Lateral connectance to floodplain areas is eliminated except at peak flows, due to altered channel morphology caused by human intervention (bank riprapping, dikes) and altered flow regime.
- ➔ All appropriate high quality spawning gravel in upstream areas is destroyed by silt deposition, preventing spawning of white suckers, leaving only mature adults. Culverting is common, contributing to impairment of fish passage
- ➔ Lack of riparian vegetation eliminates habitat for adult flying aquatic insects, reducing survival and reproduction of resident organisms and reducing successful recruitment of immigrating organisms (i.e., flight dispersal of ovipositing females).

6 **Severe changes in structure of the biotic community and major loss of ecosystem function** *Extreme changes in structure; wholesale changes in taxonomic composition; extreme alterations from normal densities and distributions; organism condition is often poor; ecosystem functions are severely altered*	**I *Historically documented, sensitive, long-lived, regionally endemic taxa*** ➔ Poor water quality, compaction of substrate, elevated temperature regime and absence of fish hosts for reproductive functions preclude the survival of any mussel fauna **II *Sensitive- rare taxa*** ➔ These taxa are absent due to poor water quality, elevated temperature regime, alteration of habitat, loss of riparian zone, etc. **III *Sensitive- ubiquitous taxa*** ➔ Absent due to above listed factors, though an occasional transient individual, usually in poor condition, may be collected. **IV *Taxa of intermediate tolerance*** ➔ Filter-feeding insects and other macroinvertebrate representatives of this group are severely reduced in density and richness, or are absent. **V *Tolerant taxa*** ➔ Low dissolved oxygen conditions preclude survival of most insect taxa except those with special adaptations to deficient oxygen conditions (e.g., *Chironomu*s) ➔ The macroinvertebrate assemblage is dominated by tolerant non-insects (Planariidae, Oligochaeta, Hirudinea, Sphaeriidae, etc.) **VI *Non-native or intentionally introduced* taxa** ➔ Native species are essentially absent ➔ Only very tolerant invasive alien fish taxa are collected (Golden shiner, Yellow perch); ➔ Number of individuals collected is abnormally low **VII *Physiological condition of long-lived organisms*** ➔ Fish biomass is very low; individuals that are collected appear to be transients and are in poor condition ➔ Incidence of parasitic infestations and disease is high; anatomical deformities and/or tumors are common ➔ Minimal evidence of recruitment or reproduction except some extremely tolerant groups may have high production; young of year age classes are absent **VIII *Ecosystem Function*** ➔ Water quality has degraded to such an extent that algal photosynthesis is negligible ➔ Decomposition of organic matter creates P/R markedly <1; the system is predominantly heterotrophic as a result of high bacterial respiration and minimal photosynthesis ➔ Reproductive success is very low ➔ Recruitment of emigrating organisms into upstream and downstream habitats is impaired due to low fecundity and high mortality rates of resident biota. **IX *Spatial and temporal extent of detrimental effects*** ➔ The *reach* and all tributaries are affected by widespread alteration of within stream conditions as a result of severely altered land-use and poor water quality. **X *Ecosystem connectance*** ➔ *Watershed*-wide land use changes and alteration of stream morphology has affected all tributaries eliminating sources of recruitment and destroying spawning habitat; ➔ Physical and chemical requirements to fulfill *life history functions* (e.g., seasonal temperature cues for mating behavior and egg development; intact nursery habitats; optimal levels of dissolved gases, etc.) are severely disrupted resulting in very low reproductive success and high mortality rates.

CHAPTER 3. HOW DO YOU DEVELOP AND CALIBRATE A BIOLOGICAL CONDITION GRADIENT?

Figure 3-1 shows the overall approach for calibrating the Biological Condition Gradient, BCG, for a specific region. This chapter discusses the technical elements and steps for calibrating a regional BCG. The calibration process includes:

- Identification of defensible biological goals (also see Chapters 1 and 5)
- Development of the conceptual foundation of the regional BCG (Section 3.1)
- Assessment and modification, if necessary, of the State's biological monitoring program to support quantitative calibration of a regional BCG (Section 3.2)
- Calibration of a regional quantitative BCG model for operational assessment (Section 3.3)

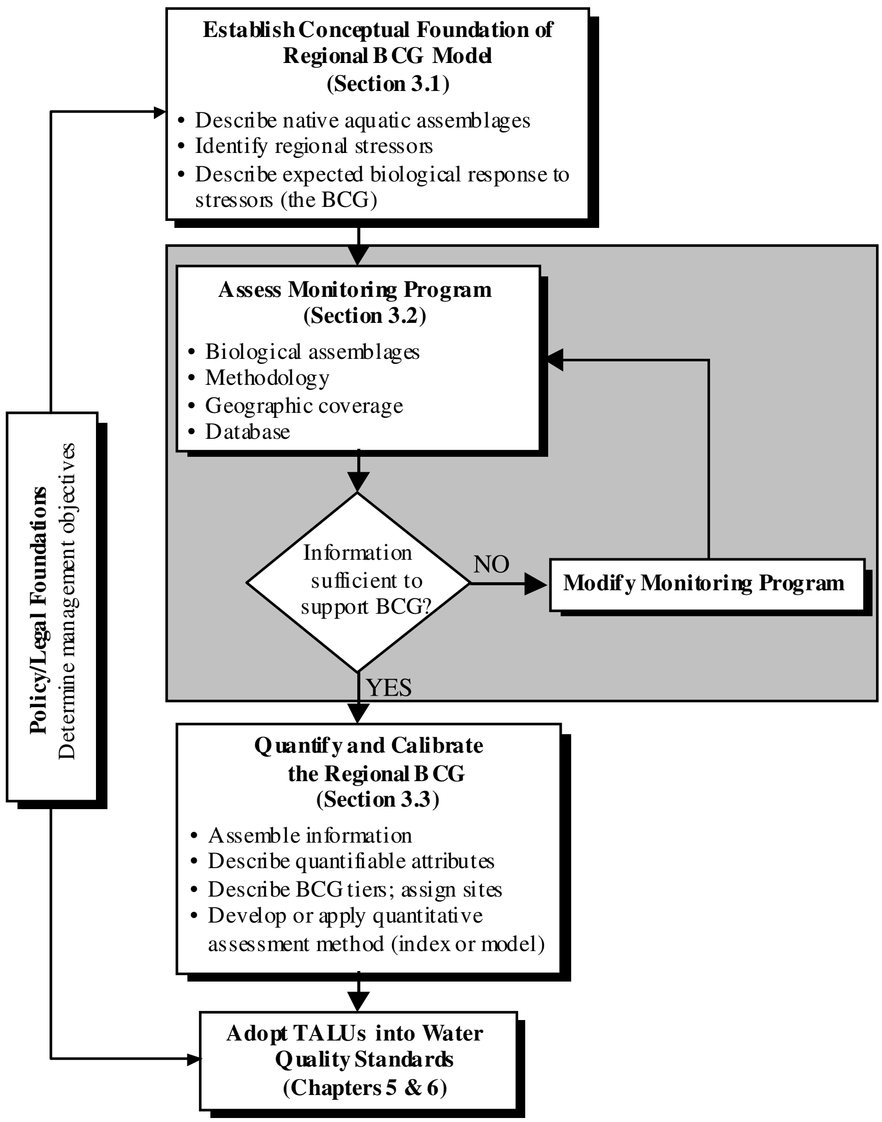


FIGURE 3-1. Technical components of the Biological Condition Gradient.

A State's water management program can support development of tiered aquatic life uses if it is flexible with respect to improvements in scientific knowledge and acknowledges that scientific advances may support adjustment of biological goals. State and Tribal designated uses form the aquatic life goals and water quality criteria (biological, chemical and physical) to protect the uses provide the basis for measuring attainment of the goals.

3.1 Conceptual foundation of a regional BCG model

The first technical component of calibrating a regional BCG is to adapt the national BCG model to regional conditions. Model development includes three components that, together, provide a complete ecological description of biological response to stressors that is consistent with ecological theory and empirical observation:

- Describe the native aquatic assemblages under natural, undisturbed conditions
- Identify the predominate regional stressors
- Describe the BCG, including the theoretical and empirically observed foundation of assemblage response to stressors

Similar to the national BCG model development process, regional BCG calibration can take place through technical panels and workshops that bring together aquatic biologists and ecologists knowledgeable about the waterbodies and assemblages in their regions. The technical experts describe native aquatic assemblages, regional stressors, and patterns of biological alteration based on both empirical observations and theoretical foundation to develop a regional biological condition gradient. The technical experts can include scientists from State and federal water quality agencies and natural resource departments, interstate river commissions, universities, and the private sector.

Expert participants in the regional model and calibration exercise should be knowledgeable about the assemblages sampled in the applicable monitoring programs (invertebrate biologists, ichthyologists, algologists, endangered species experts, etc.). The group should also include scientists involved in monitoring programs who are familiar with the sites and the organisms, plus other State, federal, university, and private sector biologists with relevant expertise. In some cases, BCGs have been initially drafted by a single experienced and knowledgeable individual, followed by a consensus process to confirm and modify the model.

3.1.1 Describe native aquatic assemblages

The BCG is grounded in natural biological assemblages that are present in ecosystems with no or minimal disturbance. Developing the BCG entails specific descriptions of the natural aquatic assemblages. The description of natural conditions requires biological knowledge of the region, classification of the natural assemblages, and, if available, historical descriptions of the habitats and assemblages.

Existing information – Information on biota in undisturbed or minimally disturbed habitats is required to develop a regional BCG model. If the State has an extensive monitoring program with undisturbed reference sites, its existing monitoring data will play an important role in developing the descriptions of reference biota. In addition to monitoring data, participants should also consult general references on biota of the region, especially references showing the historical and present-day geographic distribution of flora and fauna. These references often exist for fish and vascular plants, or may be unpublished reports and lists for threatened invertebrates such as mussels, snails, and dragonflies. However, such references are often unavailable for benthic macroinvertebrates or algae.

Classification – Developing a description of the BCG requires that biologists take into account the natural variability in assemblage structure and composition among sites and explain that variability where possible. This requires a classification system or model to predict the natural variation among sites (e.g., Wright et al. 1984, Barbour et al. 1999, Bailey et al. 2004). In this document, the term "classification" refers to identifying consistent differences between biological assemblages from undisturbed or minimally disturbed aquatic systems, if information available, and explaining those differences in terms of natural environmental gradients. Such natural gradients are encompassed within the regional descriptions of the undisturbed or minimally disturbed condition of the stressor gradient (Chapter 4).

Distributions of the organisms that make up aquatic communities are controlled by the effects of temperature, water velocity, light, oxygen, water quantity, dissolved substances (e.g., DOC, alkalinity, pH), food resources, cover, reproductive habitat, variability of physical and chemical factors, competitors, and predators. These physical and chemical factors vary geographically enabling biologists to characterize several community types by geographic location, such as cold water/warm water fish communities and low gradient/high gradient invertebrate communities. Scientists have also recognized geographic boundaries characterized by geology or vegetation (ecoregions: Omernik 1987; fish communities: Hughes and Larsen 1988; macroinvertebrate communities: Gerritsen et al. 2000). Some variables, notably measures of stream size (e.g., order, catchment area, length, total flow), have a more continuous effect on biological variables (e.g., increase of fish species richness with stream size; Karr et al. 1986).

Reference condition – Closely connected with classification of undisturbed or minimally disturbed systems and communities is the definition and measurement of reference condition. Methods for establishing reference condition need to be consistent for differing waterbody conditions to be compared (Hughes 1985, 1994; Hughes et al. 1986; Moss et al. 1987; Bailey et al. 2004; Stoddard et al. in press). Undisturbed or minimally disturbed conditions are comparable to "natural conditions," e.g. BCG tiers 1 and 2. Therefore, defining "natural" reference conditions is the starting point for development of a regional BCG. Ideally, empirical data assembled from reference sites with no or minimal levels of stressors characterize Tiers 1 and 2 of the BCG. This is because Tier 1 biological condition is, by definition, an assemblage structure, function, and taxonomic composition that is "naturally derived" from a physical environment not effected by stressors (Angermeier and Karr 1994).

Minimally disturbed sites (as defined by physical, chemical, and landscape measures) can be slightly altered from undisturbed condition, but should retain most characteristics of the resident biota in undisturbed sites. In many regions of the country where Tier 1 and Tier 2 sites may no longer exist, the reference sites used by agencies are considered "least disturbed." These sites have also been termed as the "best available," or "best existing," in the region but may be substantially altered from pristine, natural conditions. In extensively altered regions where undisturbed or minimally disturbed sites are absent, the best means to accurately characterize Tiers 1 or 2 may be through historical records of the taxonomic distributions of different assemblages and descriptions of the physical setting of undisturbed conditions (see below).

Historical descriptions – Historical descriptions help reconstruct undisturbed aquatic habitats and may help identify present-day sites that approximate historical conditions. This information is especially critical in areas where the best existing sites are significantly altered. Sources of historical information include early photographs and taxonomic collections, pre-dam and pre-irrigation physical data (USGS flow data, BLM data), and the descriptions of pioneers, naturalists, and scientists. Recent compilations and summaries of historical information have been developed where local or conservation interest is strong (e.g., Kuzelka et al. 1993, Johnson 1994). *See Case Example 3-1 on considering historical stream characteristics to estimate minimally disturbed conditions and support reference stream selections in Kansas*.

If no undisturbed or minimally disturbed reference sites exist in a region, the stressor gradient provides a means for determining the best regional candidates to act as benchmarks for comparison, i.e., "least disturbed" or "best available conditions." Chapter 4 discusses the stressor gradient and a framework to organize stressor information derived from measures of the physical, chemical, and landscape variables of a sampled site. Applying monitoring information that is organized into the stressor gradient framework will help managers evaluate the status of their waters relative to change, or departure, from reference condition.

3.1.2 Identify regional stressors

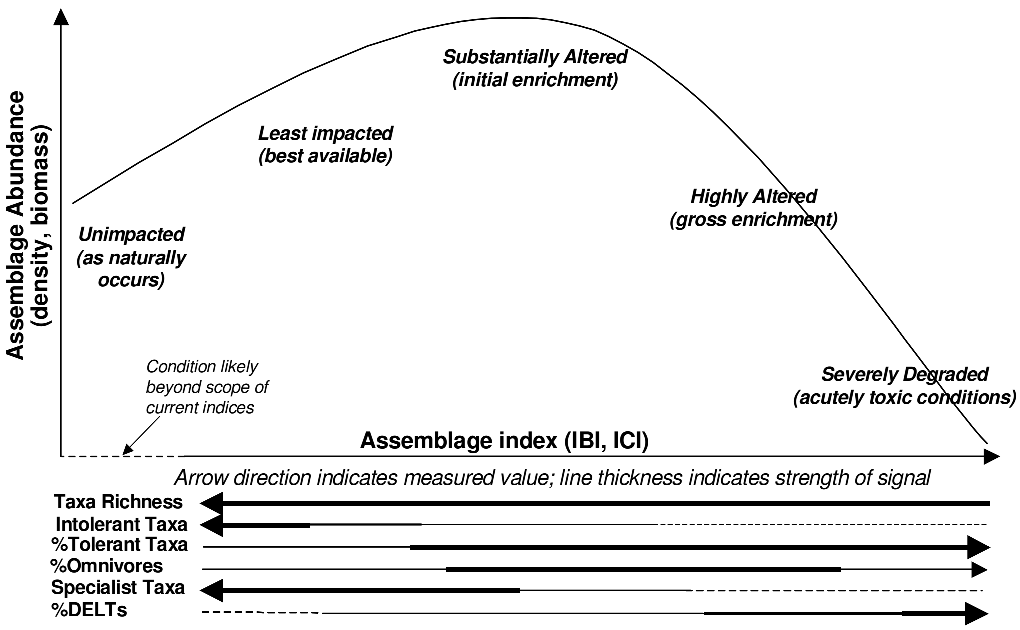
A description of regionally dominant stressors will help define expectations for biological responses that are likely to occur. This step considers sources of physical and chemical stressors and causes of landscape or habitat disturbance (the stressor gradient; Chapter 4). For example, if an ecoregion is primarily mountainous, then stressors from extensive row-crop agriculture will be relatively less frequent than stressors from other sources. Other examples of regionally important stressors include hydrologic alteration from urbanization; effluent-created permanent streams in the arid west; and acid mine drainage and related metals contamination in coal mining regions of the Appalachians and metal mining regions of the Rocky Mountains.

Identification of stressors and their sources is the first step in characterizing the stressor gradient (Chapter 4). The stressor gradient is the combination of causal factors that induce an adverse response in the aquatic biota. A conceptual model of fish and macroinvertebrate assemblage response to a regional stressor gradient ranging from undisturbed or minimally disturbed conditions to severely altered conditions was developed based on empirical observations of assemblage responses to multiple sources in Ohio (Figure 3-2). The graphic represents measured assemblage abundance (y-axis) against an assemblage index (fish IBI, macroinvertebrate ICI; x-axis) with the generalized response of selected metrics. Biological descriptions correspond to the six tiers of the BCG model and include descriptions of assemblage characteristics, chemical water quality conditions, physical habitat and flow regime, and sources of stress that are typically associated with each. This was modified from an original conceptual model by Ohio EPA (1987) and Yoder and Rankin (1995b). It demonstrates that understanding the relationship between assemblage responses and stressors is a fundamental aspect of bioassessments.

3.1.3 Describe the Biological Condition Gradient

In testing the national BCG model, regional experts calibrated it to specific regional sites and assemblages. Biologists familiar with the regions' natural aquatic communities and their responses to stress worked collaboratively to calibrate the BCG model to conditions in the following regions: Maine, Kentucky, the Central Great Plains, and selected areas in the arid west (Arizona and eastern Washington). Table 2-3 shows the resulting model for Maine.

The equivalent step in developing a regional BCG model is to develop a local counterpart to the national BCG model. The objective is to ground the BCG in local conditions. The regionally calibrated BCG describes the undisturbed or minimally disturbed aquatic ecosystems of the region, and the responses of the biota to the predominate regional stressor gradient. To the extent possible, the regional model should describe undisturbed or minimally disturbed conditions.



Biological and Stressor Gradient Descriptors

"As Naturally Occurs" ***(Pristine)***	**"Least Impacted"** ***(Exceptional)***	**"Initial Enrichment"** ***(Good)***	**"Moderate Enrichment"** ***(Fair)***	**"Gross Enrichment"** ***(Poor)***	**"Severely Altered"** ***(Very Poor)***
		Assemblage Characteristics			
Native assemblages; no symptoms of stress	"Best of what's left" assemblages; high richness; intolerants, specialists predominate	"Typical" assemblages; good richness; emerging symptoms of stress in selected metrics	"Impaired" assemblages; tolerants & generalists predominate numbers/bio-mass; loss of intolerants	"Degraded"; highly tolerant taxa pre-dominate; reduced abundance; anomalies increasing	"Severely degraded"; very low numbers; few taxa; very high % anomalies; toxic signatures
		Chemical Water Quality Conditions			
As natural; no human-made compounds present	"Best reference" quality; toxics < detection; high D.O., low nutrients	"Background reference" quality; toxics < chronic; adequate D.O., nutrients = reference	"Enriched" quality; toxics < chronic; marginal D.O. regime, nutrients > reference	"Degraded" quality; toxics > chronic; low D.O., nutrients >> reference	"Extremely poor" quality; toxics $\geq$ acute; very low D.O., nutrients >> reference; contaminated sediments
		Physical Habitat & Flow Regime			
Natural habitat and flow regime; no human-made modifications	Excellent quality habitat & flow regime; recovered from human-made modifications	Good quality habitat & flow regime; *de minimis* human modifications	Fair quality habitat & flow regime; active human modifi-cations; incomplete recovery	Poor quality habitat & flow regime; active human modifi-cations; no recovery	Severe modifi-cations; ephemeral flows; active human modifications; no recovery potential
		Examples of Sources and Activities			
No effects of human activity are evident	Point sources present, do not dominate flows; NPS impacts buffered by extensive riparian system	Point sources may dominate flows; NPS impacts buffered by good riparian zones	PS/NPS enrich-ment impacts; NPS unbufferd; channel modifi-cations; im-poundments	Gross PS/NPS enrichment impacts inc. CSOs; NPS unbufferd; chan-nel modifications; urbanization	Severe PS/NPS toxic impacts; extreme channel modifications; urbanization; acid mine drainage, severe thermal

FIGURE 3-2. Conceptual model of the response of fish and macroinvertebrate assemblages to a gradient of impacts in warmwater rivers and streams throughout Ohio (modified from Ohio EPA 1987 and Yoder and Rankin 1995b).

The BCG model may require some example data from sites to empirically ground-truth conclusions. An example regional BCG was described in Chapter 2, the Maine scenario for cold-water, high gradient streams (Table 2-3). Ohio also developed a conceptual model of the BCG, shown in Figure 3-2, as part of its tiered aquatic life use development. In addition to the description of undisturbed, natural assemblages and the predominate stressor gradient in a region, the regional model also requires a narrative description of the tiers and their biological attributes.

***A narrative description of the tiers of the BCG for the region* –** The regional model includes description of individual tiers along the gradient of biological response to stressors, including organisms present and organisms absent. The descriptions of changes in the attributes corresponding to the different tiers are derived from the consensus among technical experts as well as agreement on the number of tiers that can be discriminated across the entire gradient. The regional narrative descriptions refine the national model's descriptions of changes across the stressor gradient to reflect local conditions. (e.g., see Maine example, Table 2-3 and Ohio example, Figure 3-2). The description of the Ohio BCG is in the row titled "Assemblage Characteristics" (Figure 3-2). In Ohio, enrichment occurs at intermediate disturbance levels for the metrics (numbers or biomass).

The descriptions should account for the natural classification that applies to the region. As noted in Section 3.1.1, "classification" is defined as the process of stratifying according to natural gradients. It may be necessary to develop separate narrative descriptions for major classes of natural gradients if the biological expectations differ widely among classes. For example, the biota of low-gradient streams with fine, sandy substrates may be dominated by invertebrates adapted to those conditions, such as midges and worms. These same organisms are often indicators of degraded conditions in fast-flowing streams with coarse substrate, but may be expected to occur under the best conditions in naturally silty streams.

***A narrative description of the ecological attributes that are used to determine the tiers* –** Ecological attributes are measurable characteristics of the system (described in Chapter 2). For bioassessment programs that sample biota of target assemblages, the critical attributes are those most closely related to taxonomic information contained in the sampled assemblages. Many species can be assigned to an attribute group, and the change in the attributes is described in the conceptual model. In the Ohio example (Figure 3-2), attributes include intolerants, generalists, specialists, etc. listed in the descriptions in the first row (Assemblage Characteristics).

3.2 Data needs: Assess and modify technical program

Consistent, quality assured and controlled (qa/qc) monitoring information is key to developing a quantitative assessment system within a BCG framework. Key elements of a biological monitoring programs are listed below, correspond to design and data collection elements outlined in *Technical Guidelines: Technical Elements of a Bioassessment Program* (see Appendix C) (Barbour and Yoder unpublished manuscript). Elements of a monitoring program for quantitative calibration of the BCG are discussed below.

3.2.1 Biological assemblages

Development of a quantitative BCG can include one or more biological assemblages (e.g., benthic macroinvertebrates, fish, periphyton, phytoplankton). Choice of each of these assemblages, and field sampling methods, are discussed in *Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish* (EPA/841-B-99-002; Barbour et al. 1999).

3.2.2 Consistent methodology

Consistent and demonstrated methodology is important for calibration of a regional BCG. Methodological consistency includes sampling methods that obtain representative samples of relevant biota in the assessment unit, choice and use of sampling equipment, index period, definition of sampling site (e.g., stream reach), and allocation of sampling and subsampling effort to obtain representative estimates of composition and structure. Field sampling considerations are discussed in *Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish* (EPA/841-B-99-002; Barbour et al. 1999), and statistical considerations are discussed in *Statistical Guidance for Developing Indicators for Rivers and Streams* (Appendix E).

3.2.3 Geographic coverage

The monitoring program should have sufficient spatial and temporal coverage to provide adequate quantitative information to describe biological community expected undisturbed/minimally disturbed conditions (Section 3.1.1). This would include major geographic regions, waterbody types, and environmental gradients of pressure and stressors.

Natural Classifications –There should be sufficient reference site data in the State's database to classify natural conditions and account for natural spatial variability among sites. Classification was discussed in Section 3.1.1.

Stressor gradient – To describe the BCG, examples are used for each of the tiers that occur in the state or region. Hence, data must span the entire condition gradient from the least disturbed to the most disturbed sites in a particular region, along the entire stressor gradient.

Geographic information – In addition to routine monitoring data, geographic information helps to develop natural classification of waterbodies to refine the expected condition. As noted above, one of the requirements for developing a description of the BCG is to have a natural classification of the resource, which provides a framework for organizing and interpreting natural variability among sites. Useful geographic information includes:

- Watershed delineations – catchments of the specific sampling sites
- Physical characteristics of sampling site catchments (catchment area, distance to source, mean slope, etc.)

In addition to natural characteristics, geographic information should include information for characterizing the stressor gradient, the x-axis of the BCG and evaluating whether there are undisturbed or least disturbed reference sites. This would include information on discharges, non-point sources of pollutants, and watershed and landscape characteristics.

Reference condition – The no or low stressor end of the stressor gradient, whether undisturbed or least disturbed condition, should be well represented as reference sites and reference condition in the database. Considerations for establishing reference condition were discussed in Section 3.1.1.

3.2.4 Database

A comprehensive and complete database is critical to BCG calibration. The database should include all information collected in the monitoring program, as well as stressor and pressure information that may be collected on a geographic basis. The data must be organized and made accessible so that expert participants can easily view and interpret the data.

3.2.5 Modify monitoring program

If the specific data and information from a State monitoring program are not sufficient to support a quantitative BCG calibration, then the State may need to strengthen its technical program. Monitoring and sampling program design are not covered here. See *Technical Guidelines: Technical Elements of a Bioassessment Program* and *Statistical Guidance for Developing Indicators for Rivers and Streams* (Appendixes C and E).

3.3 Calibrate a regional BCG model

The final step in developing an assessment method using the BCG framework is to quantify and calibrate a model or system for routine assessment of waterbodies. In this step, the conceptual model that was adjusted for regional conditions is further refined and validated with data and, where possible, with quantitative relationships. The same expert panel that developed the regional conceptual model is best suited to calibrate the BCG model with quantitative information.

Regional BCG models have been calibrated for routine use in bioassessment and biocriteria programs. These calibrations can be used independently as stand-alone assessment methods, or in conjunction with existing biotic indexes. The earliest operational development took place in Maine and Ohio (Ohio EPA 1987, Courtemanch et al. 1989, Davies et al. 1995, Yoder and Rankin 1995a, Davies et al. 1999) and was the basis for the development of the national conceptual model. Regional calibration extends beyond application of the conceptual model and requires consistent operational rules so that sites can be assigned to tiers in a consistent fashion.

The following sections outline the process of regionally calibrating and developing a BCG model.

3.3.1 Assemble information

The information required to complete these tasks includes the database of consistently collected biological monitoring data from a subset of sites throughout the region and geographic and historical information where available (Section 3.2). If the State or agency has a very large data set from a long-standing monitoring program, then it is not practical to make all of the data available to the regional BCG workshop participants. Instead, select a subset of sites that represent the entire stressor gradient, from the minimally or least disturbed to the most stressed sites in the state. The objective of the rating exercise is to select a variety of representative sites across the gradient so that all tiers occurring in the region are represented in the calibration sample of sites. Some reference sites should be included in this set as well as intermediate and severely stressed sites. The data must be organized and made accessible so that expert participants can easily view and interpret the data. The following information should be available:

- A comprehensive species list for each assemblage that is monitored (e.g., macroinvertebrates, periphyton, fish), which can be sorted by higher taxonomic categories (order and family). To the extent known, tolerance values (to various stressors), trophic status (functional feeding group), habit, breeding guild, etc. should be included in each taxa list.
- Counts of abundance, by taxon, for each sample. If necessary, the database program can adjust for unequal effort among samples.
- Complete habitat data
- Field notes
- Complete field physical and chemistry data (e.g., streamflow, pH, conductivity, temperature, velocity, etc.)
- Complete laboratory chemistry results
- Landscape and hydrologic alteration of the catchments of the sampling sites, if available; otherwise land use of the smallest hydrologic accounting unit that contains the sampled catchments
- Site identification (name, ID, location)

Sites from a comprehensive monitoring program should span the range of water and habitat quality found in the state, from the best to the worst. At this point, the data will have passed QA checks and will meet the requirements for developing a BCG, outlined briefly in Section 3.2 and in Appendix C, and in greater detail in *Technical Guidelines: Technical Elements of a Bioassessment Program* (Barbour and Yoder unpublished manuscript).

Rather than expecting the expert group to work with stacks of printed data, it is useful to develop a spreadsheet that can be manipulated by participants or projected onto a screen for use during group discussions. The spreadsheet displays data from a single site at a time and calculates taxa and abundances of attribute groups. One person should be assigned responsibility for assembling all relevant data for the workshop exercise. If the State data are not well organized (i.e., not housed in a single comprehensive database), then assembling the data may require substantial time and effort.

Classification – In this stage, it may be necessary to develop, refine, or empirically test classification schemes proposed in conceptual model development (Section 3.1) if the State does not have a fully tested classification scheme for aquatic assemblages in natural waterbodies. The purpose of classification for this document was also explained in Section 3.1. Classification is influenced by the components of a monitoring design: methods, measured variables, sample size (number of sites), etc. There are several quantitative approaches to developing a classification system, including categorical models, continuous models, *a priori* methods (use of existing models), and *a posteriori* methods (empirical models using data in hand). Many references are available to help analysts develop biological classifications of waterbodies (bioassessment case studies and methods: Barbour et al. 1999, Wright 2000, Gerritsen et al. 2000, Hawkins et al. 2000, Hawkins and Vinson 2000, Smith et al. 2001, Bailey et al. 2004; textbooks: Jongman et al. 1987, Ludwig and Reynolds 1988, Legendre and Legendre 1998, Davies et al. unpublished manuscript).

3.3.2 Describe attributes

Ecological attributes are measurable characteristics of the system described in Chapter 2. These are the measures used to determine a waterbody's position along the BCG. As described in Chapter 2, attributes that are derived from taxonomic composition or organism condition (Attributes I to VII) are routinely measured and interpreted in State and Tribal water programs. As a practical matter, these are the key attributes that need to be quantitatively characterized for routine assessment.

The technical expert panel should work through the list of taxa collected in the monitoring program and assign the taxa to Attributes I through VI. In this process, the specific definitions of the attributes may be adjusted to reflect local knowledge. For example, New Jersey biologists redefined Attribute II from "sensitive-rare" taxa to "highly sensitive" taxa because rarity was not considered to be related to sensitivity to pollution, and sampling methods do not capture rare taxa with any predictable reliability. *See Case Example 3-2 for further discussion of New Jersey's tier descriptions for high and low gradient streams.*

- Attribute I consists of rare and endemic taxa, which are not often encountered by routine biological sampling methods. Their presence may be known from larger-scale surveys designed to assess rare species.
- Attributes II through V are taxonomic groupings organized according to tolerance to pollution, where Attribute II taxa are the most sensitive and Attribute V taxa are the most tolerant. These four attributes are the quantitative workhorses for assessment on the BCG and must be thoroughly characterized to calibrate a regional BCG. The tolerances of these attributes can be initially assigned based on existing tolerance estimates, but the panel should consider whether the existing tolerance estimates are accurate based on their experience and observations of the organisms.
- Attribute VI consists of introduced taxa.

Due to incomplete information, rarity in the database, or lack of knowledge, not all taxa will be assigned to an attribute.

3.3.3 Describe tiers and assign sites to tiers

Similar to the national BCG model development process, regional development can occur in workshops that bring together aquatic biologists and water quality standards experts familiar with streams in their regions. Workshop participants are asked to develop both the ecological attributes and the rules for assigning sites to tiers along the gradient. Workshops proceed as follows:

1. Participants consider the conceptual model of the BCG to identify specific biological changes that can be observed along the stressor gradient in their region. Specific metrics or attributes that can be measured within the BCG framework are identified.
2. The groups consider data from selected monitoring sites and assign the sites to tiers in the BCG based on the biological monitoring information from each site. Initially there may be disagreement among the group members, but as they become familiar with the process, sites are rated more consistently.
3. From the discussions and decisions, a set of rules is developed for assigning sites to individual tiers in the BCG.

Using the regionally adapted conceptual model (Section 3.1.3), participants examine data from selected sites throughout the region. Sites are selected from the preliminary stressor gradient (See Chapter 4) to represent the gradient as it occurs in the region. The group should consider the biological condition, species present and absent, and come to consensus on the tier to which each site should be assigned. Experience has shown that assemblages are best kept separate at this stage. The group should describe the tiers and assign sites to tiers separately for macroinvertebrates, fish, periphyton, and other assemblages.

Groups typically start with several candidate reference sites in the region in an effort to establish a reference baseline. Depending on the completeness of the database, the best sites in that database may not reflect undisturbed or minimally disturbed conditions. Additionally, if the ecoregion spans more than one state, the best sites might be in a different state or tribal land—and may not be part of the database. Ideally, calibration of the BCG in physiographic or ecological regions that cross state boundaries should be multi-state and tribal efforts. The important point here is that the best sites are *not* automatically assigned to Tiers 1 or 2. The assemblages from the candidate reference sites should be compared to the descriptions of Tier 1 and Tier 2 sites developed in the initial theoretical exercise. The following questions should be addressed:

- Do the candidate reference sites meet the theoretical expectations of Tier 1 or Tier 2? Then, if the answer is no, first validate the model's Tier 1 and 2 expectations by addressing the following questions:
    - Are these candidate reference sites minimally disturbed, that is, are there no or negligible effects from stressors?
    - Can the level of stressors be documented?
    - Is historical information available that would suggest that they are minimally disturbed?
- If these three questions are answered "yes" then the theoretical expectations and descriptions of Tiers 1 and 2 may need to be reassessed and altered. If the candidate reference sites apparently have more then than minimal or negligible levels of stressors, then they do represent examples of Tiers 1 and 2, undisturbed or minimally disturbed conditions. In many areas, sites identified as reference, especially those that are the "least disturbed," may be rated Tier 3 or even Tier 4 in the BCG.

Following development of the tier descriptions, participants continue to assign sites to tiers using the descriptions they have developed. Both the tier descriptions and the original taxa assignments may be revisited and revised in order to resolve any anomalies or issues that arise throughout the assignment process. Sites are frequently deemed intermediate (between adjacent tiers), and assigning sites to tiers does not require group unanimity. *See Case Example 3-3 on Maine's assignment of stream sites to waterbody classes (tiers) using benthic macroinvertebrate metrics*.

Tier assignments can also be tested against stressor gradients from the database. Stressor gradients (e.g., toxic metal concentrations, habitat conditions, nutrient concentrations, etc.) can be considered partial components of the stressor gradient (Chapter 4). Figure 3-3 shows an example from Ohio, showing copper concentration in the BCG tiers. In general, lower tier sites have a greater likelihood of elevated copper above the criterion level, although all tiers except the poorest (NA; very poor) included at least some sites with copper not exceeding the criterion.

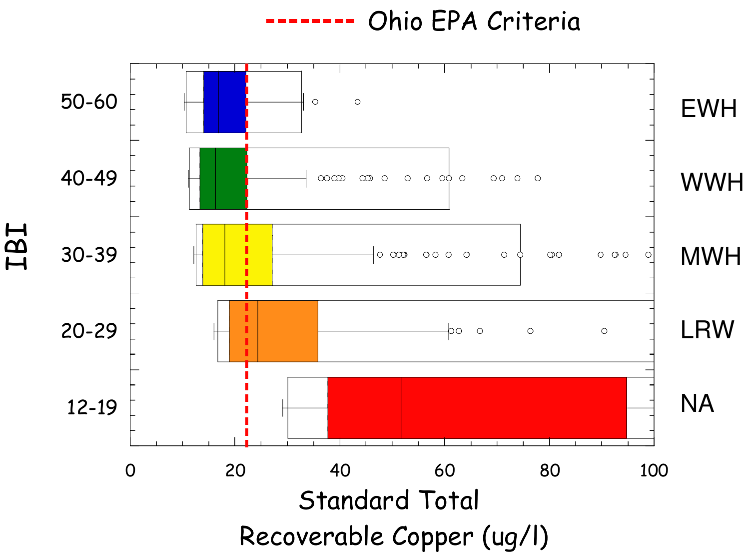


FIGURE 3-3. Ohio BCG tiers and copper concentration. Each horizontal bar approximates the tier shown on the right: EWH - exceptional warmwater habitat; WWH - warmwater habitat; MWH - modified warmwater habitat; LRW - limited resource waters; NA - non-attaining (very poor). Shaded areas are interquartile ranges of copper concentration in each BCG tier. Note that all sites in the very poor tier had copper concentration above the Ohio copper criterion (dashed line).

Setting expectations in significantly altered landscapes
In some regions, the historical conditions describing Tier 1 and 2 sites no longer exist. Many native species have been extirpated or greatly reduced, and the physical and chemical habitat of streams is completely different from the pristine, or undisturbed, condition. For example, the breaking up of native prairie sod and ongoing agricultural practices has resulted in high sediment and nutrient loads in midwestern prairie streams (e.g., Kuzelka et al. 1993). Removal of forest cover in eastern agricultural areas (e.g., Corn Belt Plains, Interior Plateau, Southeastern Plains, Riverine Lowlands) has had similar effects, although large tracts of forest cover remain or have regrown. In the western Great Plains, damming of snowmelt-fed streams and rivers has eliminated annual scouring flows and reduced sediment loads of rivers such as the Missouri, Platte, Arkansas, Rio Grande (e.g., Johnson 1994). Biological conditions comparable to Tiers 1 and 2 may no longer exist in some ecological regions of the continent. Mitigation of the resource to pristine conditions may not be currently possible *(See Case Example 3-1).*

3.3.4 Develop quantitative assessment methods

To developing a regional BCG water quality agencies should consider ecological information critically in making assessments. Biological condition tiers are narrative statements on presence, absence, abundance, and relative abundance of several groups of taxa, as well as statements on system connectivity and ecosystem attributes (e.g., production, material cycling). The statements are consensus best professional judgments based on the years of experience of many biologists in a region, and reflect accumulated biological knowledge.

Consistent application of the BCG to routine assessment and ultimately to better define designated aquatic life uses in water quality standards, will require an operational system that does not depend on reconvening the same group of experts to rate all sites. Assessments should minimize individual variability or bias, as might occur if individual assessors then interpret the rules developed by expert consensus.

Accordingly, there are a variety of ways to automate the decision tool, ranging from application of existing biotic indexes (multimetric IBI type indexes, RIVPACS indexes, BEAST applications) to development of new expert systems that specifically replicate the decision-making of the expert group that defined the BCG for the region (Appendix A; Davies et al. unpublished manuscript). Below are discussions of three methods for developing an operational assessment system, two of which use existing indexes, and the third of which develops and calibrates a system specifically for identifying tiers of the BCG. Other methods are also possible (e.g., expert systems), but the three explained below are currently used for operational bioassessment into tiers of the BCG.

Any quantitative model or procedure that is developed to assign sites to tiers should be tested with independent data that were not used to calibrate the model. This applies to all three quantitative model approaches discussed here. In general, the models are calibrated using tier assignments developed by the expert panel (Section 3.3.3). A second data set of tier assignments (also assigned by the expert panel) is then required to test the model.

Calibrating biotic indexes to the BCG

Biotic indexes such as IBIs (multimetric approaches under a variety of acronyms; Barbour et al. 1999), predictive model indexes (RIVPACS approaches; Wright 2000), and true multivariate indexes (BEAST models; Bailey et al. 2004) are all attempts to describe a biological condition gradient. As such, index approaches may be suited to identifying tiers in the gradient and for assessment in the context of the BCG.

Simple division of an index scoring range is not recommended because most indexes were not explicitly developed on a BCG framework. For example, metrics in an IBI-type index may have been selected because of strong responsiveness to stressors, rather than reflecting the conditions expressed in the BCG (see Table 2-1). If a State is to develop tiered aquatic life uses based on the national BCG model, it therefore may be necessary to recalibrate existing index models to the BCG or develop new biological models and can be used to assign sites to tiers. For example, Vermont has designated aquatic life uses as differentiated by biological threshold criteria *(See Case Example 3-4).*

Through an iterative process, scoring criteria may be developed for existing indexes that correspond with biologists' consensus on narrative descriptions of the tiers in the biological gradient. If tiers are established based on other designated uses (e.g., hydrologically modified canals), then each tier or use class can be calibrated to an index score reflecting the best potential condition for that use. Ohio used this approach to set biological criteria for four use classes (see Chapter 5).

An existing index may be calibrated to the BCG model at the level of index scores, or by deriving a new index that better reflects the BCG. Both approaches require a set of sites that have been assigned to the tiers of the BCG that were determined by the expert panel to be appropriate for the specific aquatic ecosystem (Section 3.3.3).

Calibrating index scores – The set of sites that have been assigned to tiers of the BCG are used to calibrate index scores. Index scores for the sites are examined (Figure 3-4). If separation of the index scores among tiers is good, then index thresholds can be selected to maximize the ability to discriminate among the tiers. Figure 3-4 shows a hypothetical example with five tiers (BCG Tiers 2 - 6). Separation of scores among tiers is generally good, and the solid lines indicate scoring thresholds between adjacent tiers. The exception here is that the index does not discriminate as reliably between Tiers 2 and 3 as it does between other pairs of tiers.

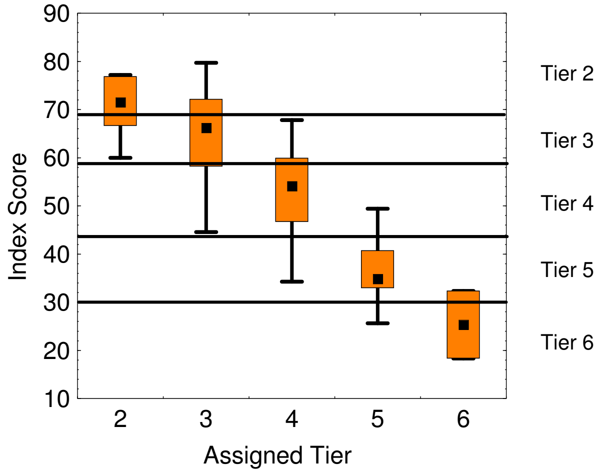


FIGURE 3-4. Hypothetical example of biotic index scores of sites assigned to BCG tiers, where the index is able to discriminate tiers most of the time. Boxes represent interquartile range of each tier; points are medians, and whiskers represent range of score outside the quartiles. Horizontal lines represent thresholds that could be applied to discriminate tiers using the index scores. In this example, there are no undisturbed reference conditions in the region.

The British Environment Agency recalibrated two RIVPACS indexes in a similar way. Initially, index scores were divided into four equal tier categories based on the statistical distribution of reference site scores (90% interval; Helmsley-Flint 2000). However, regional field biologists observed that four equal categories based on a 90% interval were insufficient to discriminate exceptional from good sites, and poor from very poor sites (Helmsley-Flint 2000). Accordingly, the indexes were recalibrated so that categories matched those determined by the regional experts. The resultant six categories are similar to the six tiers of the BCG (Table 3-4). *See Case Example 3-5 for a description of this process.*

Calibrating metrics – However, index scores may show a great deal of variation within BCG tiers, such that assigning tiers based on index scores is an inaccurate process (Figure 3-5). In the hypothetical example shown in Figure 3-5, the index is unable to discriminate among Tiers 2 through 4. In this instance, it would be necessary to revise the index to reflect tiers of the regionally calibrated BCG. Revision and recalibration of an IBI, or of other indexes, can be part of a State's routine recalibration process that occurs periodically when substantial new data have been collected.

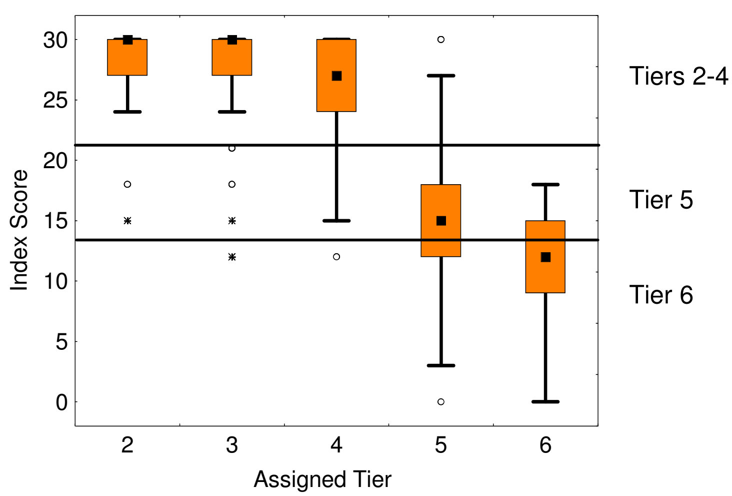


FIGURE 3-5. Hypothetical example of biotic index scores of sites assigned to BCG tiers, where the index is not able to discriminate tiers.

Model development to support BCG tiers: Discriminant model

Simple recalibration of index scores to BCG tiers may not yield distinct break-points (or benchmarks) between adjacent tiers. This is the case when sites in different tiers (as determined by the expert panel) have the same or similar index scores, showing that the index cannot discriminate among tiers of the BCG. Development of an operational tiered assessment system may require a separate index or model calibrated to the tiers.

Discriminant analysis may be used to develop a model that will divide, or discriminate, observations among two or more classes. A discriminant function model is a linear function combining the input variables. It obtains the maximum separation (discrimination) among the classes. The model is developed from a "learning" dataset where the classes have been identified. The model is then used to determine class membership of new observations where the class is unknown. Thus, a discriminant function model can be developed from a biological data set where sites have been assigned to BCG tiers. The analysis identifies variables that will discriminate among the tiers. The resultant model is then used to identify the tier to which a site should be assigned. Maine uses this method to determine whether streams are meeting biological criteria for multiple tiered uses. *See Case Example 3-6 on Maine's development of linear discriminant functions to assess tiers*.

Although it requires considerable statistical expertise to develop, the advantage of discriminant analysis is that it uses established and well-documented statistical methodology. However, it requires a relatively large set of assigned sites to calibrate the model, approximately 20 per tier. Accuracy of the model to the expert-assigned calibration and test sites can be as high as 89 - 97% (based on jack-knife tests; Davies et al. unpublished manuscript).

Using a discriminant model to develop biocriteria requires both a set of training data to develop the model and confirmation data to test the model. The training and confirmation data may be from the same biosurvey, randomly divided into two, or they may be two or more years of survey data. All sites in each data set are assigned to BCG tiers by the expert workgroup (Section 3.3.3).

One or more discriminant function models are developed from the training set to predict tier membership from biological data. Once developed, the model is applied to the confirmation data set to determine how well it can assign sites to classes using independent data not used to develop the model *(See Case Example 3-6)*. More information on discriminant analysis can be found in any textbook on multivariate statistics (e.g., Jongman et al. 1987, Ludwig and Reynolds 1988, Legendre and Legendre 1998).

Quantitative rules for tier assignments

Tier descriptions in the conceptual model tend to be rather general (e.g., "reduced richness"). To allow for consistent assignments of sites to tiers, it is necessary to operationalize, or codify, the general tier descriptions into a set of rules that anyone can follow and obtain the same tier assignments as the group of experts.

Operational rules are used to define the tier descriptions ("as naturally occur," "reduced," "greatly reduced," etc.) to quantitative or semi-quantitative rules for each attribute ("Attribute II taxa > 50% of any other attribute, ± 10%"). These rules preserve the collective professional judgment of the expert group and set the stage for the development of models that reliably assign sites to tiers without having to reconvene the same group. In essence, the rules and models capture the group's collective decision criteria.

Rule development can take place during the expert panel workshop to describe the detailed BCG and assign sites to tiers (Section 3.3.3). It requires discussion and documentation of tier assignment decisions and the reasoning behind the decisions. During this discussion, facilitators should elicit and record:

- each participant's tier decision ("vote") for the site;
- the critical or most important information for the decision – for example, the number of taxa of a certain attribute, the abundance of an attribute, the presence of indicator taxa, etc.; and
- any confounding or conflicting information and how this was resolved for the eventual decision.

See Case Example 3-7 for an example of decision rules developed during New Jersey's calibration exercise (Table 3-6).

Testing
Rule development should be iterative. Following the initial development phase, the draft rules should be tested by a group of experts to ensure that new data and new sites are assessed in the same way. This usually requires a second workshop, during which a set of test sites not used in the initial rule development and also spanning the range of stress should be assessed. Any remaining ambiguities and inconsistencies from the first iteration can also be resolved. Rules can be used directly for assessments, for calibrating one of the previous assessment methods (IBI, discriminant model), or as the basis of an expert system.

Thresholds and uncertainty
For each of the quantitative models described above, it is possible to estimate predictive uncertainties. Index variability is estimated from repeated measures at sites over one or more years, and accuracy of the quantitative model to expert consensus is estimated from the number of "correct" calls by the model. Several methods exist to estimate overall predictive uncertainty. For uncertainty of the models discussed here, see Helmsley-Flint (2000) and Davies et al. (unpublished manuscript).

Not all uncertainty is statistical, and not all issues of uncertainty can be reduced to a statistical probability. Experience with the BCG workgroups suggests that there will always be sites that fall on the border between tiers. It is important to recognize that some sites are borderline or intermediate, not that we are uncertain about where they are. This is a consequence of forcing a more-or-less continuous gradient into discrete management categories.

While thresholds between tiers do not need to reflect true discontinuities in nature, the tiers should represent detectable and consistent differences in assemblages, their taxonomic composition, and ecological function. To the extent they are consistent and detectable, they serve to inform management on how well we are protecting against degradation and making progress towards restoration goals.

Disagreement among assemblages
Once a BCG has been regionally calibrated, a possible scenario in assessment is that two assemblages collected at the same site indicate different tiers of the BCG. For example, to what tier should a site be assigned if the fish indicate Tier 2 but the macroinvertebrates indicate Tier 4? Options include:

- averaging the two assemblages (Tier 3 in this example),
- selecting the lowest assessment among the assemblages (Tier 4), or
- selecting the highest assessment among the assemblages (Tier 2).

In making this decision, it is important to consider the level of rigor in the tier assessments among the assemblages, particularly if an assessment is based on an absence, rather than presence, of information (absence of evidence is not evidence of absence). This requires considering the strength of evidence for each assemblage. Automatic calculation of an average or use of the highest assessment is neither conservative nor protective of the resource. Both Ohio EPA and the British Environment Agency have chosen to select the lowest assessment among indexes and assemblages for final tier assignments (Yoder and Rankin 1995b, Helmsley-Flint 2000).

3.4 Key points from Chapter 3

1. **The conceptual Biological Condition Gradient can be quantified and calibrated to local conditions for use in assessment and water quality criteria.** The tiers of condition described in the BCG conceptual model can be applied to local or regional conditions by regional biological experts with a sufficient monitoring database.

2. **A quantified BCG is not defined by any one monitored assemblage or methodology.** BCGs have been developed from different assemblages and methodologies (fish, benthic macroinvertebrates, artificial substrates, etc.) and by calibrating different assessment indicators to the BCG (IBI, RIVPACS, and multivariate analysis).

3. **Quantification and development of a BCG is data driven.** A regional monitoring database should be used to calibrate a BCG that meets performance requirements and QA requirements. The monitoring agency should have access to biological expertise, and should be committed to provide sustained support.

Chapter 3 has discussed transforming the conceptual scientific model of the BCG into a quantified and calibrated model for biological assessment. Chapter 4 discusses the Stressor Gradient model, the x-axis of the BCG. Chapter 5 discusses key concepts and milestones for developing tiered aquatic life uses in water quality standards that two states, Maine and Ohio, have learned based on their experience in adopting tiered uses, and is supported by their individual case histories of TALU development (Appendixes A and B). Chapter 6 presents examples of how Maine and Ohio have applied tiered uses in their water quality management program.

Chapter 3 Case Examples

CASE EXAMPLE 3-1. USING HISTORICAL INFORMATION TO IDENTIFY REFERENCE STREAMS IN KANSAS

Historical information can be used to reconstruct the pre-settlement biological baseline and estimate undisturbed or minimally disturbed conditions. Potential sources of historical data include museum fish and shellfish collections, historical notes and writings, journal entries, indigenous knowledge, published archeological studies, photographs and maps, and early biological surveys or studies.

Some knowledge of pre-settlement baseline conditions is needed when planning long-term restoration efforts in areas where undisturbed or minimally disturbed reference waterbodies no longer exist. For example, in Kansas, few streams have completely escaped the effects of large-scale agricultural and livestock practices implemented over the past 150 years. Therefore, biologists within the Kansas Department of Health and Environment (KDHE) consider available information on historical stream characteristics to estimate minimally disturbed conditions and support contemporary reference stream selections.

KDHE recognizes six general categories of aquatic biological responses to increasing levels of disturbance (Table 3-1). Class A represents natural or pre-settlement stream conditions, equivalent to Tier 1 in the BCG, in which "native structural, functional and taxonomic integrity is preserved; ecosystem function is presented within the range of natural variability." Some indication of the native character of streams in the Great Plains can be found in the narrative accounts of early nineteenth century explorers, including Lewis and Clark, Zebulon Pike, and George Sibley, among others. Railroad surveys and other investigations yielded additional information on the aquatic flora and fauna and generated maps and the earliest known photographs of many streams.

TABLE 3-1. Kansas stream biological integrity categories.

Class A:	Historical (natural) reference condition
Class B:	Contemporary (quasi-natural) reference condition
Class C:	Fully supportive of designated aquatic life use
Class D:	Partially supportive of designated aquatic life use
Class E:	Non-supportive of designated aquatic life use
Class F:	Grossly non-supportive of designated aquatic life use

Source: Kansas Dept. of Health and Environment

Although many of the biological surveys from the mid-1800s were performed after the start of intensive agriculture, they still provide valuable documentation of the occurrence of several freshwater species that soon disappeared from specific watersheds or the region as a whole. Museum collections and other historical records indicate that many creeks and smaller rivers in the Great Plains supported a variety of predominately eastern fish and shellfish species, most requiring clear water and relatively stable stream bottoms. In fact, this region was once home to more than 50 unionid mussel species. Today, several mollusca species are no longer found in most of their original habitats (Figure 3-6). Over the past 150 years, at least 11 aquatic molluscan taxa have become extinct in Kansas, and an additional 23 species are currently designated as endangered, threatened, or vulnerable.

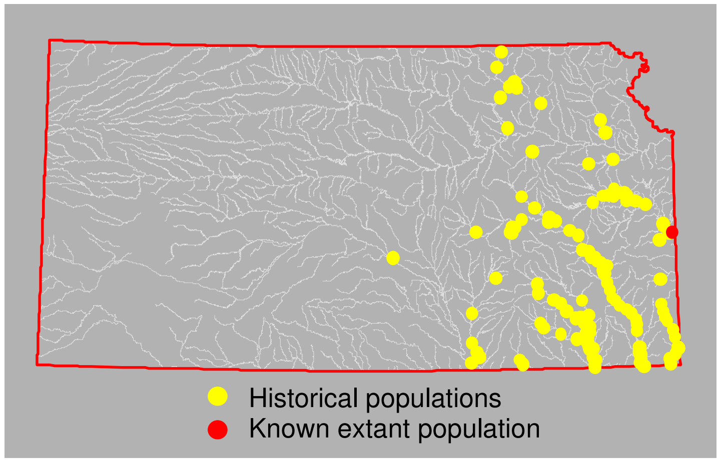


FIGURE 3-6. Decline in geographical distribution of black sandshell mussel in Kansas.

Because typical biological indexes (e.g., IBI) are usually developed from ambient "least disturbed" reference sites, they may lack sensitivity to discriminate among tiers or levels in the BCG. Surviving populations of historically occurring key species and indicator taxa can be used to further verify the minimally disturbed condition. KDHE considers historical fish, mussel, and prosobranch snail communities, and has created a "mussel loss" indicator metric that compares the taxa richness of the contemporary and historical unionid mussel assemblage for use in 305(b) and 303(d) list development (Figure 3-7). Sites retaining 90-100 percent of their pre-settlement species are deemed fully supportive of the aquatic life use, sites with 75-89 percent are considered partially supportive, and sites retaining 0-74 percent are assigned to the non-supportive category. In establishing long-term restoration goals, KDHE intends to continue drawing upon historical information sources to help ensure that the projected changes in aquatic plant and animal assemblages trend toward the pre-settlement biological condition.

There are some challenges and drawbacks when using historical data to reconstruct natural stream conditions. It takes a great deal of time and commitment to piece together numerous bits of information, especially considering the limitations and inconsistencies inherent in historical data. Much of the information is not directly comparable to modern assessment data, largely because results from previous studies and observations are often based on different sampling methodologies. Sometimes the data are not applicable because they were obtained after settlers significantly impacted the land, but often such physical habitat data are missing or incomplete. Finally, some regions settled early in the history of the nation may simply lack definitive data on the baseline biological condition.

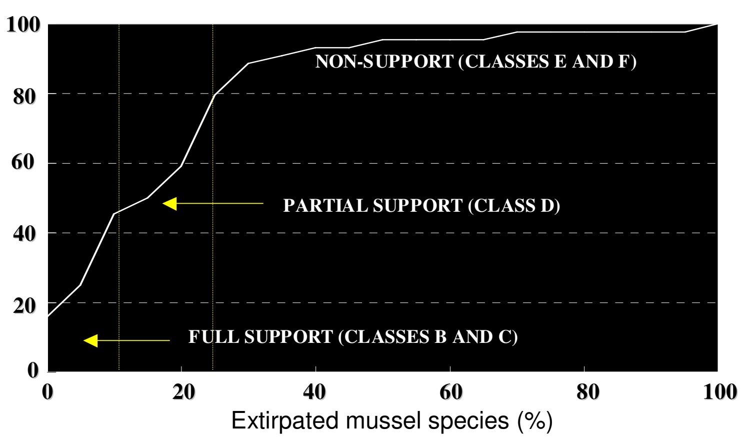


FIGURE 3-7. Cumulative frequency distribution for Kansas streams with minimum three-year period-of-record and five or more species historically.

CASE EXAMPLE 3-2. NEW JERSEY TIER DESCRIPTION

Aquatic biologists in New Jersey described tiers of the BCG for benthic macroinvertebrate assemblages of both high and low gradient streams of the state. The expert panel first assigned invertebrate taxa to Attributes I to VI. The panel redefined Attribute II from "sensitive-rare" taxa to "highly sensitive" taxa because rarity was not considered to be related to sensitivity to pollution, and sampling methods do not capture rare taxa with any predictable reliability. In addition, the panel determined that five tiers are applicable to New Jersey high gradient streams, and that four tiers describe the State's low gradient streams. For both high and low gradient streams, the panel thought that Tier 1 sites may not exist.

Table 3-2 shows the attribute matrix for high gradient streams. Attributes VII to X are not measured for the invertebrate assemblage at this time, and are not included in the matrix. The group was able to distinguish five separate tiers (Tiers 2-6) for high-gradient streams of New Jersey. The first tier described in the Maine model (Davies and Jackson in press) was not initially useful because it was not clear to the group whether Tier 1 (pristine) sites occur in New Jersey based upon benthic macroinvertebrate data alone. Other data sets (i.e. finfish communities and/or rare and endangered species) may be more useful in determining whether a site is in Tier 1. The group also determined that several indicator taxa are useful in discriminating tiers, in particular the tolerant hydropsychid caddisflies as indicators of moderate organic enrichment for Tiers 3 and 4; abundance of tubificid worms as an indicator of extreme enrichment and hypoxia for Tier 6; and complete absence of mayflies as an indicator of toxicity, also for Tier 6.

In contrast to high gradient streams, participants could only distinguish four separate tiers for low gradient streams (Tier 2, Tiers 3-4 combined, Tier 5, and Tier 6) (matrix not shown). The best-known sites in the Coastal Plain contain moderate numbers of tolerant taxa, which is a consequence of low water velocity and absence of cobble habitat rather than poor water quality. As a result, the group concluded that it was not feasible to distinguish Tier 3 from Tier 4, and combined them into a single tier.

In general, participants were able to achieve consensus on tier assignments for the sites reviewed. In some cases, there was discussion and some disagreement on which of two adjacent tiers a site should be assigned to. These intermediate sites, with characteristics of both adjacent tiers, are to be expected since ecological response to stressors is relatively continuous.

TABLE 3-2. Summary attribute matrix for New Jersey high gradient streams.

Ecological Attributes	1 Natural Condition	2 Minimal Loss	3 Some Replacement; Function Maintained	4 Notable Replacement Function Largely Maintained	5 Tolerants Dominant, Loss of Function	6 Severe Alter Structure and Function
I *Historically documented, sensitive, long-lived or regionally endemic taxa*	As predicted for natural occurrence except for global extinctions	As predicted for natural occurrence except for global extinctions	Some may be absent due to global extinction or local extirpation	Some may be absent due to global, regional or local extirpation	Usually absent	Absent

TABLE 3-2. Summary attribute matrix for New Jersey high gradient streams.

Ecological Attributes	1 Natural Condition	2 Minimal Loss	3 Some Replacement; Function Maintained	4 Notable Replacement Function Largely Maintained	5 Tolerants Dominant, Loss of Function	6 Severe Alter Structure and Function
II Highly sensitive taxa	As predicted for natural occurrence, with at most minor changes from natural densities	Virtually all are maintained and well represented (both taxa and abundance)	May be markedly diminished (in either taxa or abundance), with replace-ment by functionally equivalent *Sensitive and common* taxa	Significantly diminished (taxa and abundance)	Usually absent	Absent
III *Sensitive & common taxa*	As predicted for natural occurrence, with at most minor changes from natural densities	Present and may be increasingly abundant.	Common and abundant; relative abundance greater than *Highly Sensitive* taxa. Similar to good taxa (sensitive & common taxa).	Present with reproducing populations maintained; some replacement by functionally equivalent *taxa of intermediate tolerance.*	Frequently absent or significantly diminished (if present incidental)	Absent
IV *Taxa of intermediate tolerance*	As predicted for natural occurrence, with at most minor changes from natural densities	As naturally present at low abundances	Often evident increases in abundance	Common and often abundant; relative abundance greater than *Sensitive and common* taxa	Often exhibit excessive dominance	Richness of all taxa is low
V *Tolerant taxa*	As naturally occur, with at most minor changes from natural densities. If present, at very low abundance.	As naturally present at low abundances. May have several taxa at low abundances.	May be increases in abundance of functionally diverse tolerant taxa	May be common but do not exhibit significant dominance	Often occur in high densities and may be dominant	Usually comprise the majority of the assemblage; often either very low or very high densities.
VI *Non-native or intentionally introduced taxa*	Non-native taxa, if present, do not displace native taxa or alter native structural or functional integrity	Non-native taxa may be present, but occurrence has a non-detrimental effect on native taxa	Sensitive or intentionally introduced non-native taxa may dominate some assemblages (e.g. fish or macrophytes)	Some replacement of sensitive non-native taxa with functionally diverse assemblage of non-native taxa of intermediate tolerance	Some assemblages (e.g., fish or macrophytes) are dominated by tolerant non-native taxa	Often dominant; may be the only representative of some assemblages (e.g., plants, fish, bivalves)
XI *Potential Supplemental Attributes; Indicator taxa*	No apparent response of indicator taxa	No apparent response of indicator taxa	Initial response of indicator taxa, (e.g., increase of suspension feeders with enrichment)	Some response of indicator taxa, (e.g. increase of Caenids with silt, etc.)	Response of indicator taxa (e.g., loss of mayflies with toxic stress)	

CASE EXAMPLE 3-3. MAINE BIOLOGISTS' ASSIGNMENT OF SITES TO CLASSES (TIERS)

Maine DEP assembled a panel of three biologists to assign sites to each of Maine's three stream classes (A, B, C), and a fourth class representing non-attainment (NA). Each biologist independently reviewed biological information for each sampling event, including identities and abundances of taxa occurring in the biological sample and computed index values for the biological data (e.g. diversity, richness, EPT, etc). Physical habitat information was also reviewed including water depth, velocity, substrate composition, canopy cover, etc., in order to evaluate the effects of various habitat conditions on the structure of the macroinvertebrate community. Sample information was reviewed for the values of the given measures, relative to values for other samples in the data set. The actual classification assignment was determined by how closely the biological information conformed to the aquatic life classification standards, correcting for habitat effects. Numerical ranges, per se, were not established, *a priori*, for each measure. Instead, the information was reviewed for its compatibility with the mosaic of findings expected for each Class, listed in Table 3-3. The biologists did not have any knowledge of the actual location of the sampled sites, nor did they have knowledge of any pollution influences. Following the independent assignment of classes the biologists established a consensus classification, following an open exchange of justifications for each biologist's assignment.

Each biologist reviewed the sample data for the values of a list of measures of community structure and function. Criteria used by biologists to evaluate each measure are listed in Table 3-3.

In 64% of the cases there was unanimous agreement among the independent raters, and in an additional 34% of the samples two of the raters were in agreement and one had assigned a different classification. In three of the rated samples there was disagreement among all three raters (2%).

TABLE 3-3. Relative findings chart.

Measure of Community Structure	**Relative Findings**			
	A	**B**	**C**	**NA**
Total Abundance of Individuals	often low	often high	variable	variable: often very low or high
Abundance of Ephemeroptera	high	high	low	low to absent
Abundance of Plecoptera	highest	some present	Low to absent	Absent
Proportion of Ephemeroptera	highest	variable depending on dominance by other groups	low	zero
Proportion of Plecoptera	highest	variable depending on dominance by other groups	low	zero
Proportion of Hydropsychidae	intermediate	highest	variable	low to high
Proportion of Ephemeroptera & Plecoptera	highest	variable	Low	absent
Proportion of *Glossosoma*	highest	low to intermediate	very low to absent	absent
Proportion of *Brachycentrus*	highest	low to intermediate	very low to absent	absent
Proportion of Oligochaetes	low	low	low to moderate	highest
Proportion of Hirudinea	low	variable	variable	variable to highest
Proportion of Gastropoda	low	low	variable	variable to highest
Proportion of Chironomidae	lowest	variable depending on the dominance of other groups	highest	variable
Proportion of *Conchapelopia* & *Thienemannimyia*	lowest	low to variable	variable	variable to highest

TABLE 3-3. Relative findings chart.

Measure of Community Structure	Relative Findings			
	A	**B**	**C**	**NA**
Proportion of *Tribelos*	low to absent	low to absent	low to variable	variable to highest
Proportion of *Chironomus*	low to absent	low to absent	low to variable	variable to highest
Generic Richness	variable	highest	variable	lowest
Ephemeroptera Richness	highest	high	low	very low to absent
Plecoptera Richness	highest	variable	low to absent	absent
EPT Richness	high	highest	variable	low
Proportion Ephemeroptera Richness	highest	high	low	low to zero
Proportion Plecoptera Richness	highest	variable	low	zero
Proportion Diptera Richness	low to variable	variable	highest	variable to high
Proportion Ephemeroptera & Plecoptera Richness	highest	high	low to variable	low to absent
EPT Richness divided by Diptera Richness	high	highest	low to variable	lowest to zero
Proportion Non-EPT or Chironomid Richness	high	high	low	lowest
Percent Predators	low	low	high to variable	highest
Percent Collector, Filterers & Gatherers divided by Percent Predators & Shredders	high	highest	low	lowest
Number of Functional Feeding Groups Represented	variable	highest	variable	lowest
Shannon-Weiner Generic Diversity	low to intermediate	Highest	Variable to intermediate	lowest
Hilsenhoff Biotic Index	lowest	low	intermediate	highest

CASE EXAMPLE 3-4. VERMONT'S USE OF EXISTING BIOLOGICAL INFORMATION FOR THE BCG

Vermont used reference condition as the anchor point for assessing biological condition, and tiers of biological condition were established and described in terms of deviation from the reference condition. Biological narratives were developed, which provided guidance for evaluating degrees of deviation from the reference condition. The proposed language was intended to formalize Best Professional Judgment (BPJ) assessments by technical experts while remaining close to historical implementation. It was also critical that the new classification system maintain consistent assessment results, particularly for non-attainment findings.

Vermont tapped into more than 20 years worth of biological data collected from wadeable streams to develop biocriteria. Existing macroinvertebrate and fish assemblage monitoring data were evaluated for "reference" and "non-reference" condition in order to classify wadeable stream ecotypes and define biological reference conditions for each. Reference, or minimally disturbed, sites were determined based on BPJ. Various macroinvertebrate and fish community metrics were evaluated in order to describe their usefulness in detecting responses to disturbance.

Macroinvertebrate analysis identified four distinct wadeable stream ecotypes exhibiting unique biological characteristics: small high-gradient mountain streams; medium-sized high gradient streams and rivers; warmwater moderate gradient rivers and streams; low gradient soft bottom rivers and streams. A suite of eight macroinvertebrate community metrics was selected for the purpose of setting threshold criteria based on responsiveness to disturbance and impact. The eight metrics represent a range of structural and functional characteristics and were evaluated to minimize information redundancy. The range of reference condition was described for each metric and ecotype. Threshold criteria, based on deviation from the reference condition, were established for each ecotype consistent with the language contained in the water quality standards for each classification (Figure 3-8). Uncertainties associated with each threshold are recognized through the establishment of threshold ranges. The eight metrics are not combined into a single index number, but are evaluated separately in a BPJ analysis of use support status.

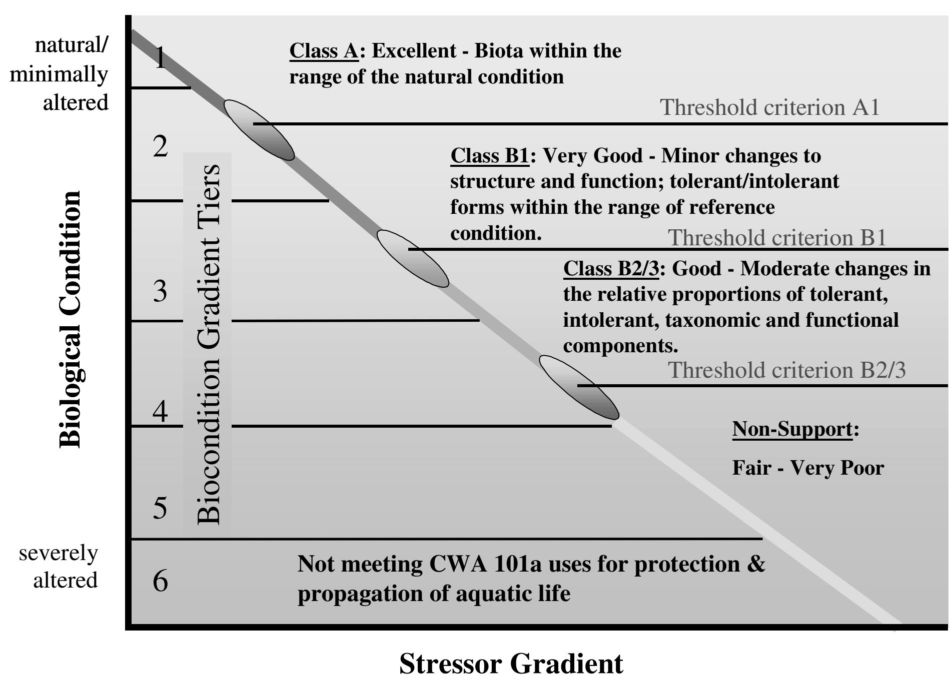


FIGURE 3-8. Vermont's designated aquatic life uses as differentiated by biological threshold criteria.

Two fish community indices of biotic integrity differentiating between strictly coldwater and mixed water assemblages were developed and calibrated to the Vermont Water Quality Standard narrative thresholds based on deviations from the reference condition. The indices combine multiple metrics representing a range of structural and functional characteristics into a single index number.

Since the BCG is continuous, it can be subdivided into any number of categories. The fish and macroinvertebrate criteria thresholds used by the Department were able to differentiate four categories of "support" status – Class A (near natural condition), high quality Class B1, general Class B2/3, and non-support (Figure 3-8). Common narrative descriptors – excellent, very good, good and fair-very poor were used to describe the thresholds. A determination of less than good was indicative of aquatic life use non-support. Categories of non-support (fair, poor, very poor) were not described.

When Vermont's new standards became effective in July 2000, all waters previously designated Class B were categorized as general Class B2/3 by default. The idea was to use the watershed planning process to propose and implement designated use reclassifications, particularly to the high quality Class B1. VtDEC is assembling candidate lists of waterbodies exhibiting high quality biological condition consistent with the Class B1 designated use. Final consideration of candidates is made via public process in order to ensure compatibility with local watershed plans and interests. Although no reclassifications have been made to date, the BCG has provided a clear visualization of the concepts of disturbance and impact, and this has been a useful tool in explaining the WQS to the public.

CASE EXAMPLE 3-5. DEVELOPING BIOLOGICAL CONDITION TIERS IN GREAT BRITAIN

In the 1980s, the Environment Agency of the United Kingdom sponsored the development of a nationwide monitoring and assessment program based on benthic macroinvertebrates. A four-year initiative, aimed at determining whether the macroinvertebrate community at a site could be predicted using physical and chemical features, led to the development of RIVPACS (River Invertebrate Prediction and Classification System). Other countries, and some states in the U.S. such as Oregon and Illinois, have subsequently integrated RIVPACS models into their biological assessment programs.

Predictive models like RIVPACS base assessments on the compositional similarity between observed and expected biota. To create a RIVPACS model for a particular region, standard protocols are followed to sample the region's biota and habitat at a network of reference sites that span the range of that region's environmental conditions. Sites are then classified based on biological similarity. Next, a multivariate model relates environmental setting (elevation, watershed area, geology) to the biological classification – this is used to estimate, or predict, the probabilities of sites belonging to biologically-defined groups and the probabilities of capturing each taxon. The current RIVPACS model, RIVPACS IIIa (Wright 2000), estimates two indexes for assessment – one based on the total number of expected taxa and a second based on expected average tolerance of the taxa. For both indexes, the model generates a list of taxa expected to occur under unstressed conditions, at greater than 50% probability for a particular assessment site. This list is then used to estimate the site's expected average tolerance value, and the probabilities are summed to generate the expected number of species. Both the number of predicted taxa that were actually observed and the tolerance value actually observed are divided by the expected values to obtain the final indexes. These indexes are compared against the model predictions to determine if the values are significantly different from the reference condition. Index values close to 1.0 indicate the site is similar to reference, and values less than 1.0 indicate deviation from reference.

Initially, the Environment Agency created four categories for the indexes – the scoring range below the 5th percentile of the index distribution of reference sites was divided into three equal categories, and the range above the 5th percentile made up the fourth. These categories, or grades, correspond to tiers of a BCG (Wright et al. 1994, Helmsley-Flint 2000). Review and application of the grades by regional biologists revealed that they did not discriminate between "good" and "very good" sites, or between "poor" and "very poor" sites (Helmsley-Flint 2000). Through cycles of data analysis and discussions with regional biologists, the Environment Agency was able to establish index thresholds for six grades, ranging from "very good" to "bad" (Table 3-4). The grades do not represent equal intervals of the index scores (Helmsley-Flint 2000). Although the British grades are determined solely by benthic macroinvertebrates, there is a distinct similarity between the narrative descriptions of the grades and the tiers of the BCG.

Assignment of a site to a grade is based on both the tolerance and total taxa indexes (Table 3-4). The indexes are independently applicable, and the lower of the two index scores determines the site grade. For example, if the total taxa index indicates "Good" but the tolerance index indicates "Fair", the site will be rated "Fair." To achieve the status of "Very Good", a site must have at least 85% of the expected taxa of an equivalent reference site and must have a tolerance index value (average score per taxon) as high as the expected value from a reference site.

Through an iterative process, the British Environment Agency was able to develop scoring criteria for existing indexes (RIVPACS N-Taxa and RIVPACS ASPT) that corresponded to regional biologists' consensus on tiers of a biological condition gradient.

TABLE 3-4. Definitions of six biological grades, developed by regional biologists of the Environment Agency in England and Wales (Helmsley-Flint 2000).

Grade	Definition	1 RIVPACS Index Scores	
		Tolerance Index (EQI ASPT)	**Taxa Index (EQI N-taxa)**
Grade a VERY GOOD	The biology is similar to (or better than) that expected for an average and unpolluted river of this size, type and location. There is a high diversity of Families, usually with several species in each. It is rare to find a dominance of any one Family.	≥ 1.0	≥ 0.85
Grade b GOOD	The biology shows minor differences from **Grade a** and falls a little short of that expected for an unpolluted river of this size, type and location. There may be a small reduction in the number of Families that are sensitive to pollution, and a moderate increase in the number of individual creatures in the Families that tolerate pollution (like worms and midges). This may indicate the first signs of organic pollution.	≥ 0.90	≥ 0.70
Grade c FAIRLY GOOD	The biology is worse than that expected for an unpolluted river of this size, type and location. Many of the sensitive Families are absent or the n umber of individual creatures is reduced, and in many cases there is a marked rise in the numbers of individual creatures in the Families that tolerate pollution.	≥ 0.77	≥ 0.55
Grade d FAIR	The biology shows big differences from that expected for an unpolluted river of this size, type and location. Sensitive Families are scarce and contain only small numbers of individual creatures. There may be a range of those Families that tolerate pollution and some of these may have high numbers of individual animals.	≥ 0.65	≥ 0.45
Grade e POOR	The biology is restricted to animals that tolerate pollution, with some Families dominant in terms of the numbers of individual creatures. Sensitive Families will be rare or absent.	≥ 0.50	≥ 0.30
Grade f BAD	The biology is limited to a small number of very tolerant families, often only worms, midge larvae, leeches, and the water hoglouse. These may be present in very high numbers. Even these may be missing if the pollution is toxic. In the very worst case there may be no life present in the river.	< 0.50	< 0.30

CASE EXAMPLE 3-6. MAINE'S USE OF LINEAR DISCRIMINANT MODELS TO ASSESS AQUATIC LIFE USE TIERS

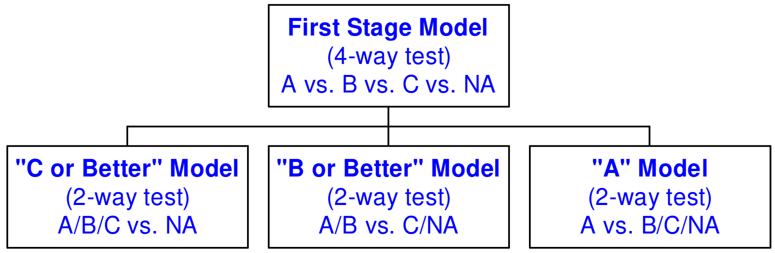
Maine identifies three aquatic life use classes for its streams – AA/A, B, and C – and also has a 4th category of non-attainment (NA) for streams that do not meet minimum water quality criteria (Table 3-5). The Maine Department of Environmental Protection (DEP) has developed a procedure using linear discriminant models (LDMs) to classify samples. LDMs are multivariate predictive models that use biological variables to determine whether a stream meets the biological criteria for classes A, B, or C, or if it falls into the category of non-attainment (Davies et al. 1995).

TABLE 3-5. Maine water quality classification system for rivers and streams, with associated biological standards (Davies et al. 1995).

Aquatic Life Use Class	Management	Biological Standard	Discriminant Class
AA	High quality water for recreation and ecological interests. No discharges or impoundments permitted.	Habitat natural and free flowing. Aquatic life as naturally occurs.	A
A	High quality water with limited human interference. Discharges restricted to noncontact process water or highly treated wastewater equal to or better than the receiving water. Impoundments allowed.	Habitat natural. Aquatic life as naturally occurs.	A and AA are indistinguishable because biota are "as naturally occurs."
B	Good quality water. Discharge of well-treated effluent with ample dilution permitted.	Habitat minimally impaired. Ambient water quality sufficient to support life stages of all indigenous aquatic species. Only nondetrimental changes in community composition allowed.	B
C	Lowest water quality. Maintains the interim goals of the Federal Clean Water Act (fishable/swimmable). Discharge of well-treated effluent permitted.	Ambient water quality sufficient to support life stages of all indigenous fish species. Change in community composition may occur but structure and function of the community must be maintained.	C
NA			Not attaining Class C

To calibrate the LDMs, stream biologists from Maine DEP assigned an initial set of streams to the four aquatic life categories: A, B, C, and NA. Assignment of samples was based on presence-absence of taxa, abundance of taxa, richness, community structure, and ecological theory. Four linear discriminant models were calibrated from the initial data set. The four models function as a two-step process to evaluate individual sites:

> Step 1: First stage model – Estimates the probability of a site's membership into each of the four classes (4-way test)
> Step 2: Second stage models – Develop more accurate membership probabilities. Each is a two-way discriminant function, which perform better than multi-way models. There are three second stage models that estimate the probabilities of membership in a given class(es) versus any lower classes (Figure 3-9).



* Aquatic life use attainment decisions are based on the three 2-way tests.

FIGURE 3-9. Series of four linear discriminant models.

The models use 31 quantitative measures of community structure, including the Hilsenhoff Biotic Index, Generic Species Richness, EPT, and EP values to classify sites. In operational assessment, monitored test sites are run through the two-step hierarchical models and assigned to one of the four categories based on the probability results. Uncertainty is expressed for intermediate sites that fall between two categories. The assessment becomes the basis for management action if a site is rated as NA, or if its assessed category (B, C, or NA; the result of the LDM) is less than the site's assigned life use class (A, B, or C). Thus, if a site was assigned life use class A, but assessment shows that it only meets life use class B or C (model assessment was B or C), then management action may be required. If a site has improved, it requires further evaluation as a candidate for reclassification to a higher class.

Maine's numeric biocriteria provide an expert system for determining attainment of aquatic life uses. The LDMs provide an empirical model for expert judgment, which in turn is ultimately derived from years of empirical observations, ecological theory, data analysis, and clearly stated aquatic life management goals. They establish a direct relationship between the model's outcomes and management objectives (the aquatic life use classes). Therefore, broad resource goals and objectives can be directly translated to scientifically defensible, quantitative thresholds (Table 3-5). The relationship is immediately viable for management and enforcement as long as the aquatic life use classes remain the same. If the classes are redefined, a complete reassignment of streams and a review of the calibration procedure would be necessary. Details of Maine's approach and statistical analysis procedures are in Shelton and Blocksom (2004) and Davies et al. (unpublished manuscript).

CASE EXAMPLE 3-7. NEW JERSEY QUANTITATIVE RULE DEVELOPMENT

After describing the BCG for high gradient streams of New Jersey (Table 3-2), the New Jersey DEP workgroup developed decision rules for assigning sites to the tiers (Table 3-6). Biologists in the New Jersey workgroup generally preferred to use taxa richness as the first and most important criterion for determining site tier assignments. Thus, the number of highly sensitive taxa was most often used to distinguish between Tier 2 and Tier 3 sites. Tier 2 should have several highly sensitive taxa (Attribute 2), but their richness may be reduced in Tier 3. For example, a preliminary rule for Tier 2 was that highly sensitive taxa richness (Attribute 2 taxa richness) should be at least 50% of the richness of any other attribute group (3 through 5). Similarly, the difference between Tiers 3 and 4 was viewed primarily as changes in the sum of richness of highly sensitive and sensitive-common taxa, such that in Tier 3 sites, the sum of taxa richness of the two sensitive groups should be at least 50% of the sum of richness of the two more tolerant groups.

Although taxa richness was generally the first criterion for the higher tiers (Tiers 2 and 3), relative abundance could override richness in extreme cases: Tier 3 was required to have more than 25% relative abundance of the two sensitive groups combined, and severely reduced abundance (< 50 organisms in the total sample, after QA determined that the sample was properly collected and processed) can downgrade a site to Tier 6 in combination with signals of potential toxicity.

Tier 5 was discriminated from Tier 4 by a significant reduction of sensitive taxa (Attributes 2 and 3) to the point where they are merely incidental if present and are not a functional part of the community. Approximately 10% relative abundance was deemed a functional part of the community. Tier 6 was discriminated from Tier 5 by increasing loss of all taxa and dominance by tolerant taxa (Attribute 5). Tier 6 could also be indicated by extreme low numbers combined with signals of toxicity (complete absence of mayflies, presence of Cricotopus), without other Attribute 5 taxa.

The rules are applied as a downward cascade: for a site to be rated as Tier 2 (the highest defined tier for New Jersey), all attributes must meet the Tier 2 condition (Table 3-6). A Tier 3 rating requires one or more failures of Tier 2 rules, but the site must meet all remaining Tier 3 rules. These rules cascade to Tier 5. Tier 6 has special rules of exceedingly low taxa richness, or abundance, or complete dominance of tolerant taxa (Attribute 5). Tier 5 consists of sites that fail Tier 4 conditions, yet also fail Tier 6.

TABLE 3-6. Proposed decision rules for New Jersey high gradient streams.

Attributes	**Tiers**					
	1	**2**	**3**	**4**	**5**	**6**
All Taxa						Low richness (<10 taxa) Low abundance (<50 individuals)
I Sensitive, regionally endemic taxa		(No rules determined for Attribute 1)				
II Highly sensitive taxa		Taxa >= 50% of any other Attribute (± 10)%	*Taxa >=2* (± 2 taxa)	May be absent	May be absent	May be absent
III Sensitive & common taxa		Taxa (2 + 3) >= Taxa (4 + 5) (± 2 taxa)	Taxa (2 + 3) > 50% of Taxa (4 + 5) (± 10%)	Taxa (2 + 3) ≥ 3 (± 2 taxa) Abund (2+3) >10%	May be absent; abund (2 + 3) <10% (or less than 3 taxa) (± 5%)	May be absent

TABLE 3-6. Proposed decision rules for New Jersey high gradient streams.

Attributes	Tiers					
	1	2	3	4	5	6
IV Taxa of intermediate tolerance					Taxa (4) ~= Taxa (5) Abund (4) >= Abund (5)	
V Tolerant taxa		<20% of total abundance (± 5%) (if tiers 2 and 3 ambiguous)	<50% of total abundance (± 10%)		High density, abundance of Attributes 4, 5	Taxa (5) > Taxa (4) (± 2 taxa) Abund (5) > Abund (4)
V.a. Indicator taxa		Tolerant Hydropsych. <= 10% abundance (± 5%)	Tolerant Hydropsych. <= 50% abundance (± 10)%		Hydropsych. may dominate Tubificidae not dominant	Mayflies absent Tubificidae dominate Attrib 5
Combinatorial rules (RxC format)		(II,2) and (III,2) and (IV,2) and (V.a,2)	(not (II,2) or not (III,2)) and (II,3) and (III,3) and ((V,3) or (V.a,3))	(Not (II,3) or Not (III,3)) and (III,4)	Not (III,4) and Not (All,6) and Not (V,6)	(All,6) or (V,6)

CHAPTER 4. THE X-AXIS: A GENERALIZED STRESSOR GRADIENT

The x-axis of the Biological Condition Gradient Model (BCG) illustrates how increasing levels of stressors in aquatic ecosystems change biological condition. This chapter presents a conceptual model that helps characterize stressor gradients by focusing on the progression from sources (changes in key environmental processes) to stressors and ultimately to their effects on biotic condition (Figure 4-1). The model also looks at the mechanisms through which these biotic components are affected. The stressor gradient model can be used to organize data and information on watershed characteristics, hydrologic modifications and stressors to thoroughly evaluate these relationships. This information will provide a foundation for States and Tribes to use the BCG to address both current conditions and ecological potential of their waterbodies, develop realistic restoration options for impaired waters, and communicate this information to the public.

4.1 The scientific foundation for the stressor gradient

Stressors affect biological assemblages and ecosystem processes both directly and indirectly, including altering metabolic pathways, energy availability and behavior of the organisms (Karr et al. 1986, Adams 1990, Poff et al. 1997). Historically, point source pollution and in-stream hydrological modifications were the dominant alterations (see 4.2.1) to fresh waters. While these issues continue today, water quality management now faces a wider variety of changes stemming from mining, forest harvest, agriculture, urbanization, industry, and even recreation (Richter et al. 1997, Bryce et al. 1999). In addition, non-contaminant related changes to aquatic ecosystem factors (see text box below) commonly impact biological conditions (Figure 1-3) and can also influence other stressors (Karr and Dudley 1981, Karr et al. 1986, Poff et al. 1997, Slivitzky 2001). Consideration of these factors and their interactions in water quality management can lead to greater improvements to biological condition than a focus on contaminants alone (Karr et al 1986).

1. **Chemical factors** (e.g., hardness, nutrients, toxic compounds)
2. **Flow regime** (including the timing and amount of water in the channel; diversions)
3. **Biotic factors** (competition, predation, disease, invading species, etc.)
4. **Energy source** (photosynthesis, inputs from land, etc.)
5. **Habitat structure** (channel shape and features, siltation, etc.) *(from Karr et al. 1986, see Figure 1-3)*

The influence of each factor on biological condition in specific waterbodies can be difficult to evaluate and quantify because each of these factors reflect both indirect and direct forces. Flow regime, for example, affects biological condition and the other in-stream factors (e.g., habitat structure, water quality) (Poff et al. 1997). Altered stream flows are associated with poor channel habitats, erosion, bank instability, and lower base flows (Poff et al. 1997). Species distributions, abundances, and competitive interactions all rely on natural flow regimes (Poff and Allan 1995, Greenburg et al. 1996, Reeves et al. 1996, Poff et al. 1997). Stream ecosystem structure and function (Vannote et al. 1980) and the riverscape concept (Ward 1998, Fausch et al. 2002) integrate the influences of all stressors. These individual and collective influences, represented by the BCG model's x-axis – the Generalized Stressor Gradient (GSG) – drive the biological condition of streams and reveal the need for a more holistic approach to stream monitoring and management. Because of the dynamic nature of aquatic ecosystems, however, all of these factors are in a state of constant flux. The natural range of conditions that native biota are adapted to may be narrow, wide, or seasonally variable, depending on the climate, topography and ecoregion in which the system occurs. A simplified model, therefore, is needed to help organize environmental factors and their relationships to stressors and biological responses.

4.2 The conceptual model for a Generalized Stressor Gradient

Building upon the Karr conceptual model, the Generalized Stressor Axis model characterizes the environmental processes and mechanisms that generate stressors which lead to biological responses within waterbodies (Figure 4-1). An event or activity that alters the aquatic system is called a **disturbance**. Ecosystems normally have some level of disturbances that characteristically occur within a range of natural variability. Disturbances beyond this range, however, can exert **pressure**[1] upon an aquatic system by altering fundamental environmental processes and ultimately generating stressors. Stressors are physical, chemical or biological factors that cause an adverse response from aquatic biota (U.S. EPA 2000b). The term "pressure" conceptually and mechanistically links larger scale landscape and hydrological disturbances with the ecological processes that are ultimately changed, leading to pressure(s) being "felt" by the aquatic biota. Stressors are what link pressures to effects on biota, via exposure mechanisms. A stressor, therefore, can be traced back to its source or tracked forward to the biological response, via a causal pathway (Figure 4-1). For example, destabilized stream banks due to removal of riparian plants could be the source of excess fine sediment to a stream. Erosion by high flows is the mechanism by which the excess fine sediments are generated, and the resulting in-stream siltation is the stressor. Smothering of bottom substrate habitat and organism gills by these fine sediments are two mechanisms by which biota are exposed and adversely affected. Invertebrate mortality and fish emigration could be some of the environmental outcomes or changes in biotic condition.

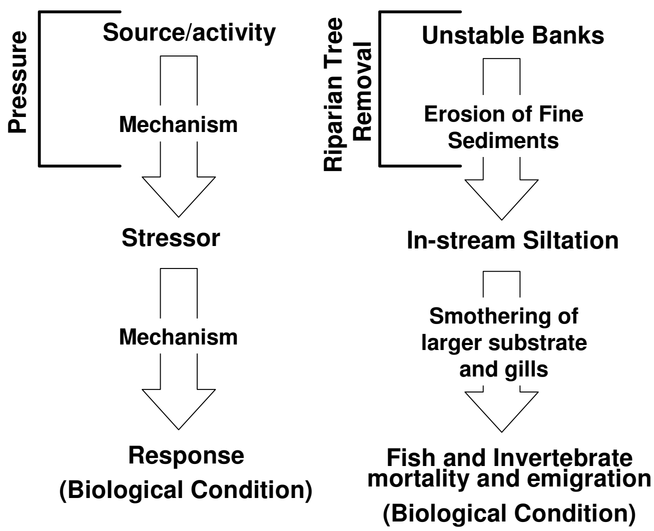


FIGURE 4-1. Conceptual model illustrating the linkages between pressure and biological condition. The specific stressor(s) and their intensity (the BCG x-axis) are created via pressure(s) acting through specific mechanisms. An example for each step of the model is also shown.

The effects of stressors on biota, however, depend on the magnitude, frequency, and duration of exposure to the stressors. Developing a BCG for a given system characterizes the general relationship between its combined stressors (the model's x-axis) and its overall biological condition (the y-axis). Multiple stressors are usually present, and thus the stressor x-axis of the BCG seeks to represent their cumulative

[1] The use of the word pressure in this context has a well-established history in the European environmental literature. Pressure is a term originally used by the European Union in its Water Framework Directive (OECD 1993). SOLEC (State of the Lakes Ecosystem Conference) also used the term pressure and defined it to be the outcomes of human activities that have the potential to cause environmental effects (Shear et al. 2005).

influence as a **Generalized Stressor Gradient (GSG)**, much as the y-axis generalizes biological condition.

4.2.1 How the model supports development of a GSG

The conceptual model provides a theoretical basis for relating single or multiple stressors to biotic responses and condition. This concept is taken further in developing a generalized stressor gradient, which, as the BCG's x-axis, is used in relating cumulative stressors to cumulative biotic effects. The factors that drive biological condition (Figure 1-3) and how condition is affected by a range of stressor intensities are used in defining the gradient. Two example GSGs are provided below.

Tables 4-1A and 4-1B outline example scenarios for humid-temperate (Table 4-1A) and arid (Table 4-1B) regions of the U.S. under differing levels of stressors. The high, medium and no/low stressor levels are used only to describe relative differences in magnitude and are not formal categories for classifying stressors. The five factors from Figure 1-3 were modified to six factors by separating toxics (e.g., copper, cadmium, mercury) from conventional chemical pollutants (e.g., nitrates, phosphorous).

When stressors are absent or low, natural or near-natural conditions of the aquatic ecosystem prevail. However, as stressors increase, one or more of the six factors can deviate from natural conditions. In humid temperate regions, for example, the loss of a watershed's forested landscape generally increases in-stream stressors by affecting flow, soil erosion, water quality and aquatic habitat structure. In arid regions, loss of riparian vegetation and cryptogamic crusts (a tightly bound mesh of lichen, algae and lower plants that prevent erosion and provide a hospitable environment for germinating plants) has the same kind of effects.

TABLE 4-1. Example scenarios for humid-temperate (A) and arid (B) regions of the US under three levels of stressors. The stressor levels are used only to describe relative differences in magnitude and are not formal categories. Karr's five factors (Figure 1-3) were modified to six factors by separating toxics from conventional chemical pollutants. These scenarios were written primarily from the reach-scale perspective, both local and watershed scale factors, however, are important for determining the condition of streams.

(A) Humid-temperate Scenario

Stressor Level	*Flow Regime*	*Habitat Structure*	*Water Quality*	*Toxics & Bioengineered Chemicals*	*Energy Source*	*Biotic Interactions*
No/Low	As naturally occurs, includes floods & low flows at natural rates and extent; High connectivity with ground water maintained	As naturally occurs, varies with size & slope, typically large wood abundant, coarse substrate, overhanging vegetation, and undercut banks are present	As naturally occurs or only minimal increase in nutrients & sediments; no point sources, includes flood turbidity & summer warming; usually cool or cold & dissolved oxygen (DO) saturated; sediments & nutrients low & pulsed seasonally	As naturally occur, typically rare and no toxics in amounts toxic to aquatic biota	As naturally occurs, varies with channel width, typically dominated by riparian woody vegetation, unless naturally autochthonous	As naturally occur, anadromy and potamodromy common or only slightly reduced by distant dams or fishing; beavers common; aliens non-detrimental; DELT anomalies absent; no or insignificant historical range changes
Medium	Flashier, increased drought frequency; some water withdrawals; low to moderate wetland drainage; dams may reduce annual floods and droughts	Reduced LWD in channel; fines slightly to moderately more abundant than expected from stream power; pool substrate moderately embedded; reduced undercut banks, overhanging vegetation, and habitat complexity; some loss of pool volume and pool/riffle proportions may be altered.	Enriched, turbidity may increase, moderate diel warming, small DO sags may occur but these rarely violate criteria; point sources minor or if they exist are treated; fish kills rare	Toxics rarely in amounts toxic to aquatic biota, but Hg may be of chronic concern to top piscivores due to bioaccumulation; sediment contamination may be detectable but not causing effects in biota.	Autochthonous production higher than expected in lower order streams; filamentous algae may be present	Altered fish age structure from fishing; beavers diminished; anomalies infrequent; sensitive aliens may dominate, tolerant aliens may be present; minor to moderate historical range reductions; cosmopolitan species may extend distributions further upstream. DELT anomalies rare; stocking may be influencing native populations
High	Flashy; highly altered drought/flood regime; mostly or entirely human controlled in urban areas; water withdrawals & impoundments if present, fundamentally alter the nature of the ecosystem	Simplified or manmade; wood, undercut banks & overhanging vegetation absent or non-functioning; rubble & trash common, substrates highly armored or embedded. Dam impoundments often present.	Highly enriched, turbid, warm; large diel DO & temperature changes; chemical and point sources inadequately treated or overwhelmed by untreated diffuse toxic pollution. Dams when present produce altered thermal regime and nutrient dynamics	Fish kills may be common in low summer flows; toxics may be present in chronic or acutely toxic amounts; bioengineered chemicals can affect growth & reproduction; high to extreme sediment contamination; fish consumption advisories serious	Mostly autochthonous or imported fine particulate organic matter or dissolved organic matter; may be too turbid for filamentous algae	Dominated by transient fishes or tolerant aliens; many historically common species extirpated; anomalies when associated with toxic impacts are abundant & serious; beavers transient or absent.

TABLE 4-1. (B) Arid Scenario

Stressor Level	***Flow Regime***	***Habitat Structure***	***Water Quality***	***Toxics & Bioengineered Chemicals***	***Energy Source***	***Biotic Interactions***
No/Low	As naturally occurs or only slightly altered, includes floods & low flows at 10-20 & 20-50 yr intervals, respectively; floods flashy; annual scouring flows; high connectivity with ground water.	As naturally occurs, varies with geology, substrate, flow, size, slope, soil, latitude, elevation & orography; relatively stable riparian vegetation, LWD in flats.	As naturally occurs with only minimal increase in nutrients & sediments, includes flood turbidity & summer warming; depending on soils, may be naturally saline or alkaline; enriched where beaver present; ash from 5-20 year fire cycles. No point sources	As naturally occur, typically rare, but may be natural sources of arsenic & selenium. No toxics in amounts toxic to aquatic biota	As naturally occurs, varies with channel width, typically dominated by riparian woody vegetation in small unconstrained channels; heterotrophic & autochthonous in wider systems	***A***s naturally occur, potamodromy (long distance river migrants) common and only slightly reduced by distant dams; beavers common; aliens absent or non-detrimental
Medium	Altered, increasingly flashy; increased drought frequency; some water withdrawals and wetland drainage; flow alterations mitigated to some extent by environmental flow releases.	Minor amounts of incision, widening or shallowing; reduced LWD in channel; fines greater than expected from stream power; bed coarsening from upstream dams; pool substrate increasingly embedded; reduced aquatic macrophytes, undercut banks, & overhanging vegetation.	Enriched, warmer & saltier, turbid at low flows, small DO sags; point sources if present with treatment. No fish kills	No acute toxicity is observed, but chronic toxicity is possible due to bioaccumulation. Fish consumption advisories likely for sensitive populations	Mostly allochthonous, but increasingly autochthonous in narrow streams; wide streams heterotrophic or autochthonous with increasing amounts of filamentous algae.	Altered fish age structure from fishing; aliens more common and beginning to reduce competitors & prey; potamodromy reduced; beavers markedly diminished.
High	Human controlled; large inter-basin transfers; ground water overdrawn; effluent dominated streams below cities. Highly altered drought/flood regime; droughts yield more dry channels; withdrawals & dams severely alter nature of the ecosystem.	Largely manmade; little or no LWD, undercut banks & overhanging vegetation; highly sedimented; construction rubble & trash common.	Highly enriched, turbid, warm; large diel DO changes; most point sources inadequately treated or overwhelmed by untreated diffuse toxic pollution; dams produce altered thermal regime.	Fish kills in low summer flows or after rains; toxics seasonally or always present in acutely toxic amounts; mine spills; bioengineered chemicals affect growth & reproduction.	Mostly autochthonous or imported fine particulate or dissolved organic matter; filamentous algae common if turbidity allows it.	Dominated by transient fishes or tolerant aliens; fish consumption advisories serious; once-common species now threatened, endangered or extirpated from large portions of their historical ranges; potamodromy rare and erratic; beavers transient or absent.

4.3 How the BCG model and management actions are linked

Pressure, as used in this document, applies to the environmental processes that can be altered by certain activities and the mechanisms from those activities that generate stressors. Many landscape altering activities can be quantified with such measures as population density, proportion of land devoted to agriculture, total miles of roadway, or quantities of water used /released. These activities, however, may or may not generate stressors. Actions can be taken that insulate stream processes from the environmental pressure of certain activities, helping to maintain or restore the ecological potential of an aquatic system.

Controls and Best Management Practices (BMPs) are management actions designed to mitigate or reduce the levels and effects of stressors that adversely alter stream ecosystem function. BMPs can function in a number of ways: they may reduce the stressors being generated by sources, reduce the exposure of biota to stressors, or increase the resistance of an aquatic ecosystem to adverse changes. For example, urbanization without controlling for the effects of added impervious surface is a pressure that often results in reduced biological condition. The typical alteration of water flow (such as more frequent flooding due to increased runoff) causes stressors. The mechanism for flow alteration is the creation of large expanses of impervious surfaces, characteristic of most cities. Impervious surface speeds up the flow of water over the land during rain events often resulting in more frequent and more intense floods. Constructing retention ponds to store run-off water is a control measure that doesn't alter the pressure of urbanization, but may reduce the stressors acting on the stream system. Mechanistic processes operate between pressures and stressors, and between stressors and biological response (Figure 4-1). Understanding these mechanisms, and how they operate, is the key to identifying the likely effect of a particular management action and its likelihood to produce the desired response in biological condition. In the retention ponds example above, the pressure (urbanization) and mechanism for stressor generation (excessive surface run-off) still exist, but their influence on in-stream stressors has been neutralized by a management action, and therefore the exposure mechanism influencing the biological community was reduced or eliminated.

The basis of the BCG model is that increased pressures can generate increased stressors, and in turn, increased stressors are associated with decreasing biological condition (Figure 4-2A through D). Systems that are minimally affected by stressors exhibit natural condition (Tables 4-1A and 4-1B). Human activities may exert pressure and generate stressors on aquatic systems, resulting in changes from the natural state. Typically, the stressors on aquatic systems increase as pressures increase (Figure 4-2A dashed line). Effective management practices, however, can alter the effects of pressures and reduce stressors. The solid, curved line in Figure 4-2B represents this theoretical relationship graphically. With effective controls and/or BMPs, a given amount of pressure (vertical fine dashed arrow rising from the pressure axis) results in a lower stressor level (where the dashed arrow intersects the stressor axis). Figure 4-2B illustrates the influence of effective management in changing the pressure/stressor relationship in ways that will subsequently improve the biological condition.

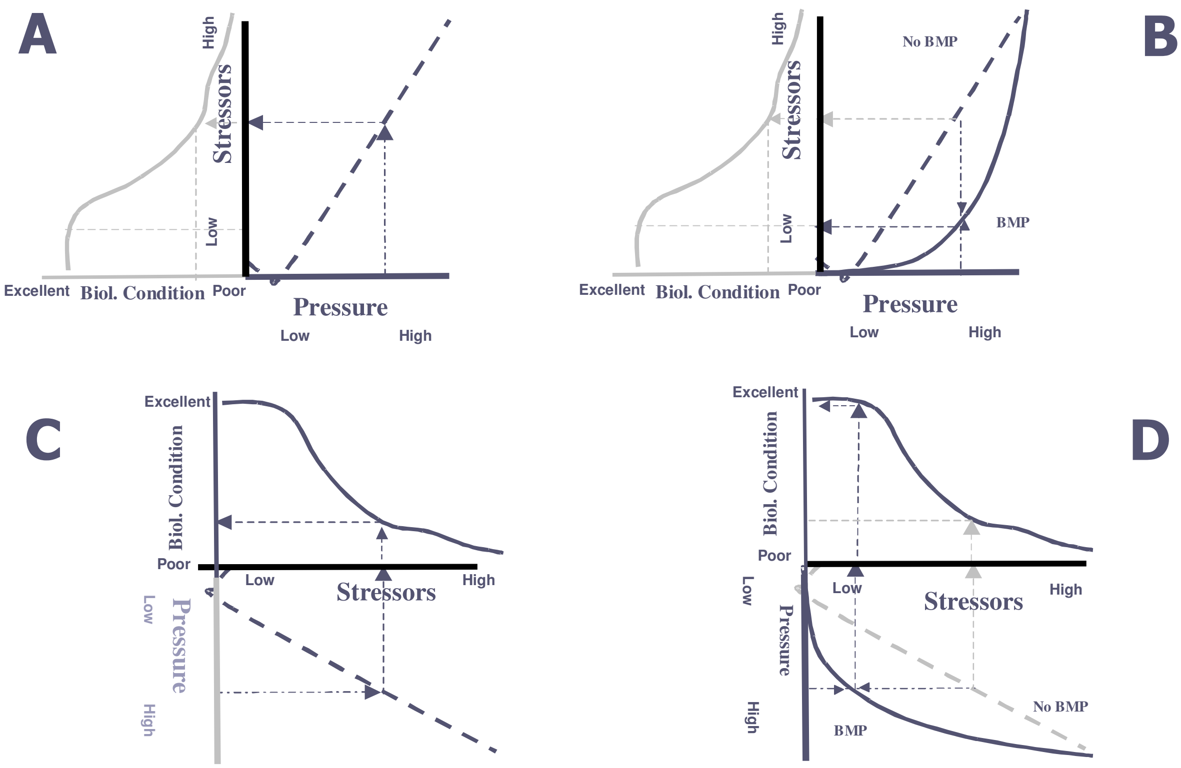


FIGURE 4-2. Relationship between pressure, stressors, and biological response.

Figure 4-2C is a 90 degrees clockwise rotation of Figure 4-2A. Stressors (which are shown to increase in response to increasing pressure in Figure 4-2A) are now on the x-axis. Biological condition is shown as the response variable on the y-axis. This represents the biological condition-stressor relationship developed in Chapter 1. In this example, the moderate-high effect of the stressors (dashed arrow rising from the stressor axis) results in poor biological condition (the point where the dashed arrow intersects the biological condition axis).

Figure 4-2D shows Figure 4-2B rotated 90 degrees clockwise. As in Figure 4-2C, stressors are on the x-axis, and biological condition is shown as the response variable on the y-axis. The effect of low levels of stressors (dashed arrow rising from the stressors axis in Figure 4-2D) results in near excellent biological condition (where the dashed arrow intersects the biological condition axis). The pressure-stressor relationship has been shaded out. But it reminds us how, together, pressure and management actions (i.e., permit limits, BMPs, channel restructuring) can determine stressor levels, and ultimately, the condition of the biota. The specific effects of stressors on biological responses will depend on the type, magnitude, duration, and frequency with which the stressor occurs. These stressor attributes are, in turn, a result of the cumulative pressures exerted on the ecosystem and relevant management decisions to mitigate these pressures.

Different types of disturbances can exert pressure on an ecosystem through altering fundamental processes such as water flow, transport of materials, watershed/riparian structural dynamics, channel structural dynamics and biological activities. For example, dams and impoundments alter flow, natural biological activities and material transport by creating lake conditions in a stream environment, and creating barriers to fish movements and migration. Sediment, nutrient and organic matter transport are all reduced downstream of impoundments and water quality attributes such as natural temperature fluctuations and dissolved oxygen are often altered by dams. When severe enough, these alterations act as stressors to the downstream community.

Tools can be developed that characterize the relationships among pressures, altered processes, the stressors they generate, and the resulting biological responses. Information from pressure and stressor indicators provides insight on how changes in these fundamental processes may be affecting the biological condition of water resources (Table 4-2). Understanding how specific stressors are generated and the influence of specific stressors on biological condition, provides the underpinnings for the BCG's stressor axis. Further, it reveals potential opportunities for management actions to reduce stressors and counteract the alteration of fundamental processes.

TABLE 4-2. Fundamental environmental processes typically altered by disturbances that ultimately generate stressors. For each process, example mechanisms that link pressures (pressure indicators) to stressors and typical stressor indicators for the major environmental factors impacted are listed. Management Actions (indicated by the grey bar place holder), when effective, will alter the effect of the pressures, thereby alleviating some or all of the mechanism for increasing stressors.

Process	**Pressure Indicators** (potential for stress)	**Mechanism for Stressor Production**		**Stressor Indicators for each Major Factor**	**Comments**
Flow alteration	-% impervious area -road density -% urban -population-density -storm sewer miles -# diversions -# of dams -point source dischargers	-acceleration of water flow -reduced groundwater infiltration -increased peak flow -more frequent elevated events -reduced base flow -suspended fines during floods increased subsequent deposition -increased incision and changes to channel structure due to increased power during flood	Management Actions (BMPs /Controls)	<u>Flow regime</u> -changes in flood frequency -changes in base flow -changes in drought frequency -changes in stream power	Alteration of flow includes changes to the rate, volume and timing of discharge Alteration of water flow also changes materials transported and channel structure; the consequences also affect water quality, toxics, and energy sources (See Table 4-1) Alterations of habitat lead to changes in fish life histories adapted to natural flow regimes leading to migrators, rheophils and nonguarding lithophils & lithopelagophils being replaced by residents, generalists, and polyphils
				<u>Habitat Structure</u> -increased fines -increased armouring of substrate	
				<u>Water Quality</u> no direct indicators	
				<u>Toxics</u> no direct indicators	
				<u>Energy Source</u> no direct indicators	
				<u>Biotic Interactions</u> -lose certain fish life histories	

TABLE 4-2. Fundamental environmental processes typically altered by disturbances that ultimately generate stressors.

Process	**Pressure Indicators** (potential for stress)	**Mechanism for Stressor Production**		**Stressor Indicators for each Major Factor**	**Comments**
Alteration of materials transported	-point source dischargers and discharge constituent levels -km of riparian buffers -% impervious surfaces -road density -row crops -population-density -atmospheric deposition -CAFOS -km^2 of tile drains -non-point sources -logging -# mines -# septic systems -fertilizer use -irrigation -# quarries	-erosion of surface solutes, sediments and warmer water -increased discharge of chemicals from point sources -increased algal biomass results in greater daytime photosynthesis and night-time respiration -increased material input adds carbon and nutrients that increase biological activity	Management Actions (BMPs /Controls)	Flow regime See above Habitat Structure See above Water Quality -altered nutrient concentrations -altered TSS -altered TDS -altered temperature -altered turbidity -increased coliform Toxics -increased contaminants in fish -increased pesticides -increased bioaccumulative organics -increased metals in water and sediments Energy Source -suspended algae could increase or decrease Biotic Interactions -changes in invertebrate functional groups to more filterers or grazers -sensitive species diminished; % tolerant species increased -algal community changes	Sediment deposition alters habitat complexity and structure; during floods, remobilization of buried materials can reintroduce nutrients and bioaccumulative toxics to the food web Algal growth may increase due to increased nutrients or decrease due to high turbidity. These state changes will affect invertebrate functional group composition (e.g. more filterers due to high levels of suspended organic matter or more grazers if alterations in material transport are reduced).

TABLE 4-2. Fundamental environmental processes typically altered by disturbances that ultimately generate stressors.

Process	**Pressure Indicators** (potential for stress)	**Mechanism for Stressor Production**		**Stressor Indicators for each Major Factor**	**Comments**
Changes to channel structure	-km channelized -# dams -# culverts -density of road crossings -valley fills -diversions -levees -bank-stabilization -riprap/ Concrete -floodplain losses -snagging of LWD	-flow alteration -solute sediment transport -direct engineering activities	Management Actions (BMPs /Controls)	Flow regime -changes in mean velocity -changes in discharge Habitat Structure -reductions in number of habitat types -reduced pool depth -reduced substrate heterogeneity -increased embeddedness -changes in width: depth ratio -reduced number of snags -reduced woody debris -reduced off channel habitat (including braiding index) -reduced flood plain connectivity, bank angle/stability Water Quality no direct indicators Toxics no direct indicators Energy Source -autochthonous allochthonous shift Biotic Interactions -reduced pool dwelling organisms -loss of specialized insectivores -blocked migrations -reduced spawning habitat	Alteration of normal channel shape and depth alters the dissipation and flow of energy during hydrological transport both laterally and longitudinally which ultimately reduces habitat complexity and therefore reduces community diversity.

TABLE 4-2. Fundamental environmental processes typically altered by disturbances that ultimately generate stressors.

Process	**Pressure Indicators** (potential for stress)	**Mechanism for Stressor Production**		**Stressor Indicators for each Major Factor**	**Comments**
Changes to watershed/ riparian structure	-% of area developed -# trapping permits -levees -tile number-drains/ditches -km of streamside roads -surface area of off stream ponds or wetlands -valley bottom grazing -riparian width -riparian continuity -aggregate mining -devegetation -fragmentation	-landscape alterations -clearing of vegetation -paving surfaces -building structures -agricultural practices -draining or filling wetlands -riparian disturbance or removal -increased algal biomass results in greater daytime photosynthesis and night-time respiration -increased material input adds carbon and nutrients that increase biological activity	Management Actions (BMPs /Controls)	<u>Flow regime</u> -changes in flood severity and response time <u>Habitat Structure</u> -reductions in amount of woody debris -loss of riparian trees -reduced riparian structural complexity -reduced off channel habitat (including braiding index) -reduced flood plain connectivity -increased riparian fragmentation <u>Water Quality</u> -increased fine -increased sediments -increased nutrients -increased ions <u>Toxics</u> no direct indicators <u>Energy Source</u> -P/R changes -increased algal contributions; -shift from allochthonous toward autochthonous <u>Biotic Interactions</u> -loss of sensitive species -addition of intermediate species and total species -changes in invertebrate functional groups to more filterers or grazers -algal community changes	Watershed/riparian bottom and vegetation provides hydrological assimilative capacity during flooding and contributes water and nutrients during interflood periods; these are also important nursery habitats and refugia that maintain biodiversity

TABLE 4-2. Fundamental environmental processes typically altered by disturbances that ultimately generate stressors.

Process	Pressure Indicators (potential for stress)	Mechanism for Stressor Production		Stressor Indicators for each Major Factor	Comments
Changes in biological activity	-# non-indigenous invasive species (NIS) -# fish stocked (species individuals) -baitfish sales -fishing licenses -creel census results -# Guide licenses	-stocking programs -foraging by invasive species (grazing, predation) -habitat modification by invasive plants and fish	Management Actions (BMPs /Controls)	Flow regime -macrophytes (decreases-> accelerate flow; increases->reduce flow)	Loss of species from invasives and stocking practices
				Habitat Structure -macrophytes/algalmats alter substrate	
				Water Quality -increased BOD	
				Toxics no direct indicators	
				Energy Source no direct indicators	
				Biotic Interactions -NIS (native gamefish decline, hatchery fish increase) -native fish/benthos and riparian vegetation and birds increasingly replaced by aliens -sensitive specialists replaced by tolerant generalists (birds, fish, invertebrates, plants) -increased fish diseases and anomalies -riparian vegetation may be eliminated	

4.3.1 Additional considerations for the stressor axis

The concepts of spatial and temporal scale are critical issues in adequately defining a stressor axis. Stressors may be introduced through diffuse or point sources delivered from upstream in the channel or watershed, or laterally from riparian, floodplain or upland sources. Pollutants can also be delivered through atmospheric sources from above, or below from groundwater sources. Activities in the watershed or along the waterbody corridor will influence the connectivity and integrity of the water resource. Stressors are expressed over temporal and spatial scales ranging from a one-time, localized event to chronic exposures occurring continuously over vast landscapes. Pressures, stressors, and responses operate at different spatial and temporal scales (Figure 4-3). These are not independent of one another in either space or time; therefore, consideration of multiple pressures is essential. An additional consideration is that any given pressure creates multiple stressors, which in turn affect biological condition. The steady accumulation of small pressures in watersheds results in "cumulative impacts," which present added challenges for characterizing, evaluating, and managing stressors.

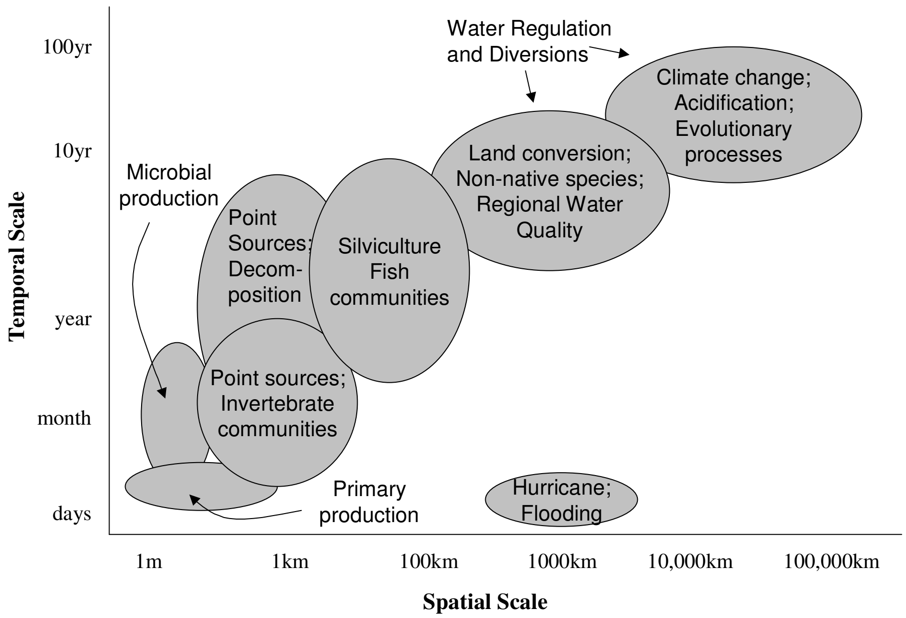


FIGURE 4-3. Perspective of scale for pressure-stressor-response variables (modified from Richards, C. and L.B. Johnson. 1998. Landscape perspectives on ecological risk assessment. In *Risk Assessment: Logic and Measurement*, M.C. Newman and C. Strojan (eds.). Ann Arbor Press.).

The complexity of the relationships between biological condition and stressors at various spatial and temporal scales, underscores the importance of using sound information to identify and link these stressors back to the pressures that cause them. To a large degree, this is the critical step in gaining stakeholder support for restoration and protection actions as well as for changes in activities or behaviors. As discussed earlier, fine sediment is commonly identified as a stressor across the United States because of the smothering of important habitat. Identifying the relative contributions of various sources of these sediments is more challenging (e.g., bank erosion, upland erosion, spatial sources), but also critical to remediation efforts.

4.4 How a GSG can be developed and calibrated

Developing and calibrating a stressor gradient must be based on appropriately classifying aquatic resources and establishing reference conditions or other scientifically defensible approaches. Classification (e.g., biogeographic regions, basins, biological considerations) is a critical first step so that the temporal and spatial scales of the dominant stressor categories and sources can be addressed (Herlihy et al. in press, VanSickle and Hughes 2000, McCormick et al. 2000, Waite et al. 2000). Of equal importance is establishing the appropriate reference condition for a particular area (Hughes 1985, 1994; Hughes et al. 1986; Moss et al. 1987; Stoddard et al. in press), because that is the benchmark against which areas to be evaluated will be compared (as discussed in BCG Section 3.1.1).

Like the biological condition axis, the stressor axis is anchored in the natural, or undisturbed or minimally disturbed, condition (i.e., Tier 1 BCG). However, reference may represent minimally-disturbed (i.e., nearly natural) or least-disturbed (i.e., best available) conditions depending on the level of disturbance that exists across the geographic area of interest (Stoddard et al. in press, Hughes 1994). Linking regional factors, pressures, and stressors with biological condition into a BCG will assist States and Tribes in identifying levels of disturbance and the primary drivers of biological condition in their watersheds. If no undisturbed or minimally disturbed reference sites exist in a region, a stressor axis provides a means for determining the best condition or regional candidates to act as benchmarks for comparison, i.e., "least disturbed" or "best available conditions." The stressor axis concept will enable managers to place the status of their stream ecosystems into a regional context and prioritize actions. The reference condition approach, which describes the potential biological condition of the region's waters, provides a framework to set appropriate restoration endpoints for that resource and region.

The next step involves quantification of in-stream stressors, riparian condition, landscape characteristics and riverscape alterations, as well as point source discharges and other localized pressures. Calibrating stressors along natural gradients (waterbody size, catchment area, stream power, elevation, latitude, and geology) can improve ability to detect pressure effects by removing the confounding effects of stressor gradients with natural gradients (Fausch et al. 1984, Hughes et al. 2004, Kaufmann and Hughes in press). There have been many efforts to characterize pressures and incorporate quantitative information into environmental assessment programs (Table 4-3). Riparian condition has been widely recognized as affecting the physical habitat and biological condition of streams (Naiman and Decamps 1990, Fitzpatrick et al. 2001, Lammert and Allan 1999, and Lattin et al. 2004). In some circumstances, watershed condition was more important (Roth et al. 1996, Snyder et al. 2003). Wilhelm et al. (unpublished manuscript) used both catchment (i.e., watershed) and riparian disturbance for the development of their non-wadeable habitat index for streams in Michigan. Wang and others (in press) found that fish assemblages were most influenced by local environmental factors in largely undisturbed catchments. However, as the level of catchment disturbance increased, the importance of catchment-scale factors increased and that of local-scale factors decreased. These studies indicate how important regional and local factors are for determining the relationship among sources, stressors, and biological condition and the most appropriate scale for addressing these relationships.

TABLE 4-3. Percent variance in biological response (R^2) explained by catchment and riparian land use, and percent land use producing poor IBI scores (modified from Hughes et al. unpublished manuscript).

Authors	Response Variable	R^2 Catchment	R^2 Riparian	N	Location & % Land Use for "Poor" rating
Bryce & Hughes (2002)	Fish IBI	0.40	--------	13	OR/ 50% urban
	Fish IBI	0.35	--------	16	Appalachia/ 15% urban
	Diatom IBI	0.29-0.36	--------	16	App./ declines w/ ag.
	Benthos IBI	0.48-0.67	--------	16	App./ 50% ag., 20% mined
Fitzpatrick et al. (2001)	Fish IBI	0.31	0.58	25	WI/ 70% ag.
	Diatom IBI	0.16	ns	25	WI/ag.
	Benthos IBI	ns	ns	25	WI/ag.
Hughes et al. (unpublished)	Fish IBI	0.42	0.38	104	OR/ rd. density >1.9 km/km2
Karr & Chu (2000)	Benthos IBI	0.25	---------	66	WA/ 40% impervious
Klauda et al. (1998)	Fish IBI	0.68	---------	61	MD/ 60% urban
Lammert & Allan (1999)	Fish IBI	0.01	0.22-0.28	18	MI/ declines w/ riparian ag.
	Benthos IBI	ns	ns	18	MI/ag.
Lattin et al. (2004)	Fish IBI	ns	0.20-0.46	25	OR/20% network riparian ag.
Leonard & Orth (1986)	Fish IBI	0.60	---------	44	WV/ rd. density >1.7 km/km2
McCormick et al. (2001)	Fish IBI	0.05-.08	---------	313	App./ declines as deforested
Mebane et al. (2003)	Fish IBI	0.45-0.56	---------	41	OR/ 25% deforested
	Fish IBI	0.56	---------	30	ID/ 15% irrigated ag.
Morley & Karr (2002)	Benthos IBI	0.53	0.00-0.82	34	WA/ 45% impervious
Roth et al. (1996)	Fish IBI	0.50	0.02-0.38	21	MI/ 80% ag.
Snyder et al. (in press)	Fish IBI	0.16-0.64	0.02-0.17	20	WV/ 15% urban
Steedman (1988)	Fish IBI	0.64	0.67	10	ONT/ 95% ag., 60% urban
Wang et al. (1997)	Fish IBI	0.48	---------	134	WI/ declines w/ deforesting
Wang et al. (2000)	Fish IBI	0.34	---------	47	WI/ 5% impervious
Wang et al. (2001)	Fish IBI	0.04-0.31	0.26-0.34	47	WI/ 5% impervious
Yoder et al. (2000)	Fish IBI	0.41	---------	101	OH/ 30% urban

Once the suite of stressors and pressures are measured or quantified for a given group of waterbodies, the next step is to determine if more than one stressor gradient exists and how they are related (i.e., are there several gradients based on different pressures, activities or landscapes?). Dealing with these multiple stressors and pressures can be complicated. A direct multiple correlation approach was taken by EMAP in the mid-Appalachian Highlands where poor quality streams were most often associated with alien fish, channel sedimentation, and riparian habitat alteration out of several hundred possible stressors (U.S. EPA 2000a). Kaufmann and Hughes (in press) used correlation and multiple linear regression analyses to determine that low stream IBI values were associated with excess streambed fines, bed instability, higher water temperature, higher dissolved nutrient concentrations, and lack of deep pools and cover complexity. These stressors were most strongly associated with riparian disturbance and road density. Effects were more pronounced in streams draining erodible sedimentary bedrock than in those draining more resistant volcanic terrain. States and Tribes could use similar multivariate approaches for identifying the stressor(s) most associated with measures of biological condition in their regions

A method employed in the Great Lakes Environmental Indicators (GLEI) project to characterize disturbance to the U.S. Great Lakes coastal region, used principle components analysis to reduce over 200 GIS variables into a single gradient (Danz et al. 2005). The GLEI approach individually considered six different kinds of disturbance: agriculture, atmospheric deposition, land cover, human population, point sources, and shoreline alteration. A watershed-based approach was used to reflect the premise that the environmental effects of these activities in coastal watersheds can influence environmental conditions in (downstream) coastal ecosystems. The first principle component from their analysis explained 73% of the variance in the agriculture variables and was interpreted as an overall gradient in stressors across the basin (Figure 4-4). Environmental responses such as water quality, fish assemblage metrics, and bird abundances were strongly correlated with this stressor gradient.

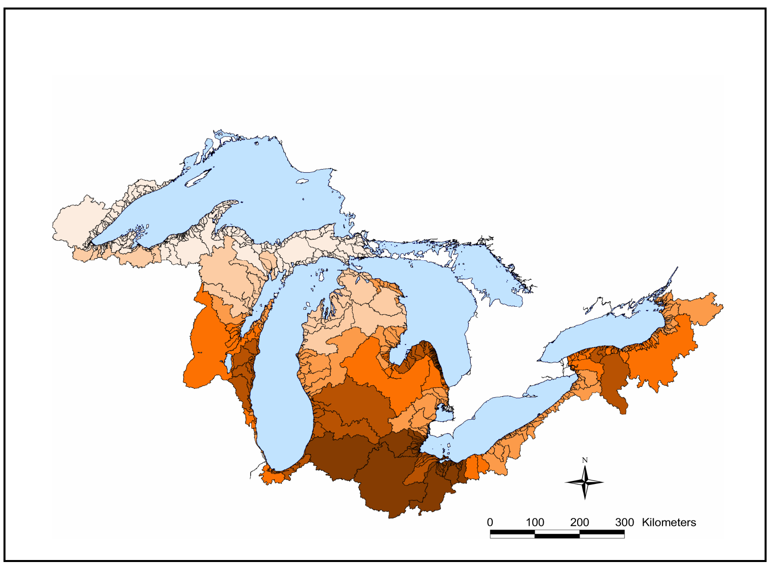


FIGURE 4-4. The first principal component of the agricultural variables for the U.S. Great Lakes basin. Darker shading indicates greater amounts of agriculture.

When multiple sources and stressors interacted to form the stressor gradient for a given watershed, GLEI found it desirable to develop a visual display of PCA axis 1 that subsumes the multiple stressors by portraying a single disturbance gradient. While the pressure-stressor model could eventually be developed and visualized as a single gradient from low to high levels of stressors (Figure 4-4), different individual and combinations of stressors are expected to dominate in different regions. Furthermore, the depiction of individual categories of stress provides important information about potential mechanisms affecting the state of the system. The GLEI researchers created a flow diagram (Figure 4-5) that details their steps for quantifying a stressor gradient (modified from Danz et al. 2005).

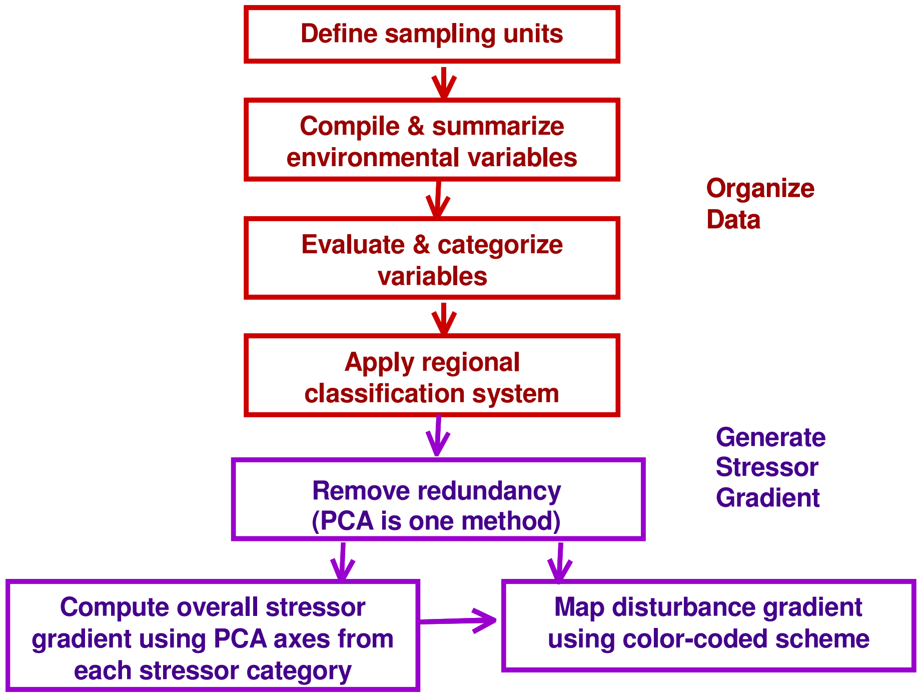


FIGURE 4-5. Flow diagram detailing the steps used by GLEI researchers in quantifying their stressor gradient (modified from Danz et al. 2005).

Whether using a single or multiple stressor gradient, all this information needs to be assembled to develop a model that integrates the components of pressures and establishes a baseline for using stressors to interpret biological responses. Relationship models that describe the associations among stressors, the processes that generate them, and biological conditions (responses) need to be developed. If possible, the extent of management actions (e.g., controls/BMPs) needs to be identified and ways to characterize these actions need to be considered (although this is an area of active research). The degree of deviation from natural conditions and the types of stressors present will affect restoration potential and therefore BMP effectiveness. Examples of tools that are currently available for characterizing a suite of pressures are: Analytical Tools Interface for Landscape Assessments (ATtiLA), National Land Cover Database (NLCD), and air photos.

Calibrating a stressor axis depends on the scale of the question to be addressed. The stressor axis should be developed independently of the biological information to avoid circularity when developing the BCG. In the development of their non-wadeable habitat index (NWHI), Wilhelm et al. (unpublished manuscript) used catchment and riparian disturbance gradients (CDG and RDG respectively) to select and weight habitat metrics at both watershed and reach scales. While the final NWHI was strongly correlated to disturbance measures and included habitat metrics that supported this relationship, a true test of the relationship between their stream response measure and disturbance measures would require a new, independent data set. The GLEI researchers used a wide range of publicly available data sets to quantify five different classes of disturbances. Their stressor axis is currently being calibrated. Stressor development and calibration involves using sufficient information to characterize relative positions along the axis and, in particular, being able to anchor the upper end (i.e., low or no stressors) and the lower end (i.e., high level of stressors) (Whittier et al. in press). This can be accomplished via a combination of public consensus, best professional judgment, and empirical approaches (e.g., Areas Of Concern (AOC), Great Lakes Environmental Indicators (GLEI) approach, and index development) (Whittier et al. in press, Danz et al. 2005, U.S. EPA 2000b).

4.5 Key points from Chapter 4

1. The stressor gradient provides a framework for organizing and interpreting information about watershed characteristics and using those characteristics to predict aquatic ecosystem biological responses. It helps us understand the observed biological conditions and the stressors related to those conditions. It can help identify the predominant stressors affecting the aquatic biota and develop effective management actions to mitigate their effects.

2. Understanding how specific stressors are generated and how they affect biotic condition provides the underpinnings for the BCG's stressor axis and ultimately the basis for interpreting the influence of stressors on biological condition.

Incorporating Tiered Aquatic Life Uses Into State and Tribal WQS: Case Examples

Tiered aquatic life uses are descriptive narratives of designated uses that are supported with numeric biocriteria and chemical and physical criteria.

As a key component of State and Tribal water quality standards, designated uses define the goals for a waterbody, determine the criteria to protect it, guide management outputs, and, ultimately, environmental outcomes. Aquatic life tiers couple descriptive narratives (tiered uses) with supporting numeric criteria. The specificity of designated uses greatly influences the level of precision at which a water quality management program operates. Incorporating tiered aquatic life uses into water quality standards can have a positive effect on water quality management outcomes. States that have made this transition have demonstrated that tiered aquatic life uses promote both the development of more appropriate aquatic life use goals and biological criteria to measure attainment of those goals. The data and experience developed from tiered uses supported by comprehensive monitoring have multiple uses in the water quality based approach to pollution control (Figure 5-1).

The preceding chapters of this document describe ways of better characterizing and defining the biological and physical condition of waterbodies and their aquatic life uses. These next two chapters discuss the underlying principles and processes involved in developing tiered aquatic life uses and applying them in water quality management based on "lessons learned" from State experiences. Maine and Ohio are two States that have adopted tiered aquatic life uses in their WQS and have implemented them through systematic monitoring and assessment. The experiences of Maine and Ohio provide a sequence of steps, or milestones, that can serve as a template for other States to follow. These milestones are:

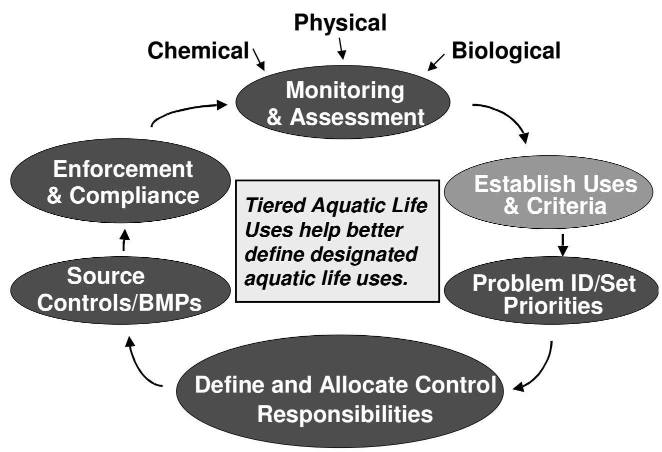


FIGURE 5-1. U.S. EPA Water Quality Based Approach to Pollution Control based on Chapter 7, Water Quality Standards Handbook.

1. **Establish conceptual foundation**
2. **Merge scientific and policy foundations**
3. **Establish monitoring program**
4. **Develop and validate quantitative thresholds**
5. **Apply tiered uses in water quality management**

Both States developed tiered aquatic life uses for similar reasons: 1) to incorporate ecologically relevant outcomes in goal setting; 2) to guide cost-effective, defensible management decisions; 3) to measure incremental progress in meeting management goals; and 4) to merge the design and practice of monitoring and assessment with the development and implementation of WQS. Chapter 5 captures the "lessons learned" by Maine and Ohio in their development of tiered uses (Milestones 1 – 4) and Chapter 6 presents case examples about how each State has benefited from this approach (Milestone 5).

CHAPTER 5. KEY CONCEPTS AND MILESTONES IN THE DEVELOPMENT OF TIERED AQUATIC LIFE USES

Tiered aquatic life uses should be derived based on knowledge of the aquatic biota (specifically the assemblages used in biological assessments) and the factors that determine their distribution, abundance, and composition. Tiered narrative statements and numeric biological criteria can represent measurable benchmarks along a regionally calibrated biological condition gradient (BCG). Maine and Ohio have determined that these benchmarks represent attainable conditions in their States for the protection or restoration of surface waters through implementation of WQS. Since waterbodies are assigned to tiered uses based on a comprehensive ecological database (biological, chemical, physical assessments), Maine and Ohio are more confident that both the uses themselves and any changes to a waterbody's condition are ecologically relevant.

5.1 Key concepts for developing tiered aquatic life uses

Maine and Ohio's tiered aquatic life uses represent the goals for individual waterbodies. Their tiered uses share the following common characteristics:

- uses are ecologically-based
- uses include the structural and functional properties of the specific aquatic communities that inhabit an aquatic ecosystem
- attainment is based on measurable biological criteria, which are indexed to a regionally relevant reference condition
- implementation integrates monitoring, assessment, and WQS

These characteristics are discussed more fully in the Maine and Ohio case histories (Appendixes A and B). But, two key concepts that Maine and Ohio have learned are:

1. Tiered Aquatic Life Uses Should Be Ecologically Based

Tiered uses should be built on a strong ecological foundation that provides a credible basis for the protection and restoration of aquatic resources. Tiered uses should reflect the collective attributes of the BCG and encompass the structural and functional attributes and processes of an aquatic ecosystem. As discussed previously, Figure 1-4 illustrates how pollution leads to exposures and responses, both ecological and human health, which can affect the status of waterbody-specific designated uses. Because the designated use is initially stated in narrative and qualitative terms, the challenge is to logically and appropriately relate the chemical, physical, and biological criteria to the designated use. The more precise the statement of the designated use, the more accurate the associated criteria can be as an indicator of that use.

Linking tiered uses to a regionally calibrated BCG provides the scientific framework for determining the biological condition and potential of individual waterbodies, which is the basis for assigning the appropriate tier. Tiered use narratives should include explicit references to the protection of aquatic life and specify the structural and functional properties that are to be protected. The derivation and calibration of numeric biocriteria should assure ecological relevance consistent with the properties of the regional aquatic fauna.

2. Linkage of Tiered Uses to the BCG via Biocriteria

The BCG attributes are incorporated directly into tiered aquatic life uses through biological assessments and with biological criteria. The development of biocriteria is an important part of the process in accomplishing this task and should adhere to the technical components of the overall TALU process

(Appendixes C, D, and E). Karr et al. (1986) recommended six key elements in the development of bioassessment tools and biocriteria:

1) measure(s) must be biological
2) measure(s) should be interpretable at different trophic levels and provide a connection to other organisms and assemblages not included in the biological assessment process
3) measure(s) must be sensitive to the environmental conditions being assessed
4) response range must be suitable for the intended application, i.e., encompassing the full range of the BCG
5) measure(s) must be reproducible and precise within acceptable limits for data collected over space and through time
6) variability of the measure(s) must be low enough to detect changes along the entirety of the BCG

Representative indicator assemblages are used to measure attainment of the biocriteria as part of the derivation process. As such, biocriteria represent the measurable ecological properties of a tiered aquatic life use.

5.2 Key milestones for developing tiered aquatic life uses

The Maine and Ohio case histories (Appendixes A and B) reveal conceptually consistent, but technically different ways of developing tiered uses including numeric biological criteria and a comprehensive monitoring and assessment program. However, the process followed by each demonstrates common tasks and milestones that States and Tribes can use as a template for developing tiered uses. These milestones and tasks are illustrated in Table 5-1 and consist of five major steps:

Milestone 1. Establish Conceptual Foundation *(Maine and Ohio Case Histories, part I)*

- Establish an interdisciplinary, collaborative approach to the development of tiered uses (ecological, technical, and legal)
- Identify and acquire appropriate staff and management expertise

Milestone 2. Merge Scientific & Policy Foundations *(Maine and Ohio Case Histories, part II)*

- Link management objectives with technical program
- Evaluate for consistency with existing water quality standards framework
- Draft or refine narrative aquatic life use descriptions

Milestone 3. Establish Monitoring Program *(Maine and Ohio Case Histories, part III)*

- Develop methods and monitoring design, establish reference conditions, build baseline database and database management system
- Logistics: staffing, facilities, and equipment

Milestone 4. Develop/Validate Quantitative Thresholds *(Maine and Ohio Case Histories, part IV)*

- Program implementation: develop biocriteria and water quality program support (initiating the process of using TALUs and biological assessments to support water quality management tasks)
- Validate the accuracy of ecological expectations with empirical data
- Program maintenance: refine biocriteria and maintain water quality program support (maintaining the process of using TALUs and biological assessments including the continuous evaluation of tools, criteria, and processes based on what is being learned via a systematic approach to monitoring and assessment; includes expansion to other aquatic ecotypes)

Milestone 5. Application in Water Quality Management *(Chapter 6; Maine Case History, part IV)*

- Apply biocriteria to support WQS
- Integrate tiered biocriteria with other types of chemical and physical criteria

Milestones 1 - 4 describe the key tasks in the development of tiered aquatic life uses. Milestone 5 addresses the application of tiered uses in water quality management. Ideally, the milestones can be accomplished sequentially, each laying the appropriate scientific or policy foundation for the next step. However, many States will have already accomplished some or even a majority of the tasks, particularly under Milestone 3 (Establish Monitoring Program). Some may also use biological assessments for support functions beyond status assessments, but perhaps lack the formal tiered use framework in their WQS or have remaining technical development issues. Maine and Ohio found that capacity for conducting biological assessments is an equally important issue and generally included 5-10% of State water quality management program resources. They found that this level of funding should make available sufficient resources to carry out the development, maintenance, and assessment tasks on a statewide basis.

Table 5-1 and Figure 5-2 include many of the major tasks in the development of a program and they can serve as a "road map" to determine where a particular State program stands regarding the goal of developing and applying tiered uses in its water management programs. Figure 5-2 can also be used as a guide for identifying, prioritizing, and organizing outstanding and remaining tasks. Furthermore, there is a transition under Milestone 4 from an emphasis on development of a tiered aquatic life use approach to program maintenance. Program maintenance includes ongoing evaluation and "fine tuning" of the bioassessment tools and criteria as the program matures. It also includes the further development and refinement of assessment and management tools and criteria as data, experience, and knowledge are gained via systematic monitoring and assessment. Maine and Ohio initially developed tiered uses and biocriteria for streams and wadeable rivers and currently either have developed or are evaluating tiered uses and biocriteria for other waterbody types (e.g. nonwadeable rivers, wetlands, lakes and estuaries). Program maintenance can also include the development of tiered uses for these other types of waterbodies. Evaluating whether there is a need to change existing use designations for specific waterbodies is another important task. This is accomplished during the triennial review process with decisions based directly on outcomes from systematic watershed monitoring and assessment and historic data.

Milestones 1 - 4 and Figure 5-2 reflect a sequence of strategic steps in the development of tiered aquatic life uses. A functional and effective program will emerge if essential theoretical, technical, and legal elements are addressed and fully integrated throughout the development process. Table 5-1 shows typical tasks associated with each founding element and the type of professional expertise required to accomplish them. One of the key "lessons learned" in Maine and Ohio is that problems arise when technical and management activities are done in isolation from each other. A collaborative and interdisciplinary approach that blends technical and management activities yields better decisions at all levels.

The triennial review process is readily adaptable to developing and then refining uses on a watershed basis, and to making needed adjustments to bioassessment tools and criteria. As the program develops and matures over time, and as resources become available, application of a tiered use framework can advance from condition assessment to formal incorporation into water quality standards.

TABLE 5-1. Expertise and tasks for key TALU milestones.

Conceptual Foundations	Technical Foundations	Policy/Legal Foundations
	Professional Expertise Required	
✓ Senior professional biologists ✓ Regional ecological experts	✓ Professional biologists ✓ Taxonomists ✓ Field support staff ✓ Statistician ✓ Database managers	***Initial concept formulation:*** ✓ Senior professional biologists ✓ WQS managers ***Later stages***: All of the above plus… ✓ Senior management ✓ State legal counsel ✓ Legislature or WQS board ✓ Stakeholders
Milestones 1, 2 and 4	**Milestones 3 and 4**	**Milestones 1, 2 and 4**
	Essential Elements	
▪ Literature review of stress ecology studies for locale ▪ Develop regional BCG model ▪ Determine expected biological assemblage response to typical stressor scenarios; ▪ Identify ecological attributes necessary to maintain a functioning ecosystem (to help establish goals for protection or restoration)	▪ Clarify classification issues (confounding natural gradients of locale); ▪ Define reference conditions ▪ Determine monitoring approach and strategy ▪ Exploratory data analyses to validate/refine BCG model ▪ Best available, best tested metrics to assess status of ecological attributes of interest ▪ Set thresholds that correspond to BCG tiers, that protect essential ecological attributes	▪ Determine management objectives; ▪ Identify priority aquatic resources ▪ Cross-walk BCG to WQS context- (how good a fit is provisional BCG/TALU conceptual model to existing use classes and WQ criteria) ▪ Seek early review of the legal standing of any proposed changes to WQS- strengthen and clarify language ▪ Account for public values and economic constraints/realities

Based on the commonalities between Maine and Ohio's experiences, several important "lessons learned" were identified for States and Tribes that are considering developing tiered aquatic life uses.

- **Interdisciplinary approach to development:** Development of tiered aquatic life uses is most successful when active cooperation and close working relationships exist among the individuals charged with technical/scientific development and oversight of water quality standards.
- **Plan enough to be certain of success… and use adaptive management approach:** Clear knowledge of scientific and legal principles should guide every step of planning and development. An adaptive management approach is beneficial throughout the development process because new technical information and management understanding are gained as part of the process. An adaptive management approach incorporates needed flexibility into a program by building on the new knowledge and insights.
- **"Proper" sequencing versus logical decisions:** The exact sequence of developmental events is not as critical as the necessity of following a plan that is logical for a particular State or Tribe, builds on current program strengths and reflects rigorous adherence to scientifically and legally sound foundations.
- **Graduated application to support water quality management decisions:** Some level of condition assessment and regulatory decision-making (application in water management) can happen as soon as a credible monitoring program is established and linked to narrative TALU goal statements.

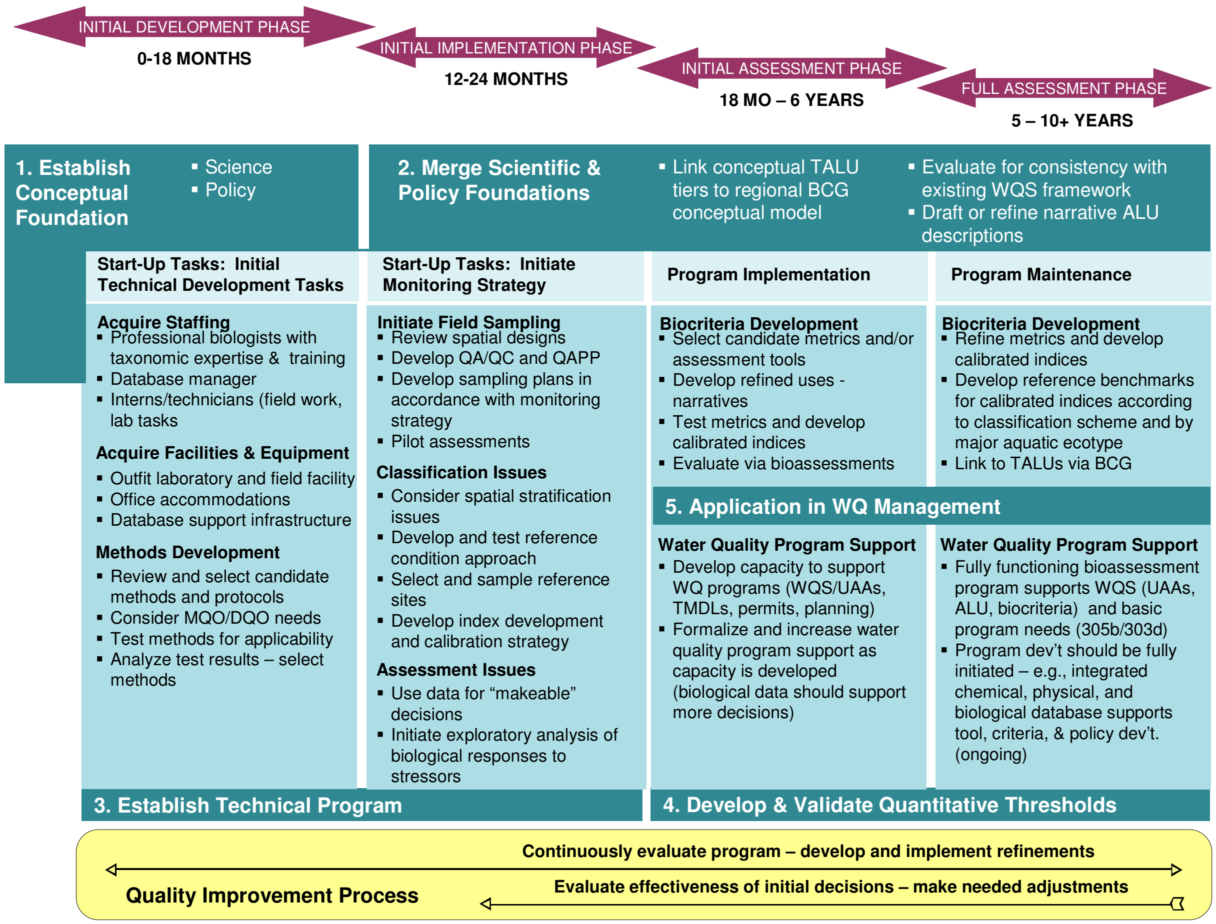


FIGURE 5-2. TALU and biocriteria program development tasks: Timeline and key milestones. A process of sequential tasks and milestones that States can follow in the development and implementation of tiered aquatic life uses and attendant biological criteria.

5.3 Using TALUs to support water quality management

The adoption of tiered uses should positively influence water quality management outputs and outcomes. Tiered uses in State and Tribal water quality standards, coupled with a systematic and comprehensive monitoring and assessment program, can provide an essential link among a wide variety of water quality management programs. In Maine and Ohio, the end result have supported baseline CWA management programs such as NPDES permitting, construction grants, and, more recently, the revolving loan program, basin planning (including TMDLs, listings of impaired waters, development of restoration plans), and nonpoint source assessment. The comprehensive support of water quality management that emerges from systematic monitoring and tiered aquatic life uses in Maine and Ohio is made possible by following the milestones shown in Table 5-1 and Figure 5-2 to establish and develop a program. Monitoring supports day-to-day water quality management needs and can take place at multiple scales including a statewide, regional, watershed, or site-specific basis.

A sustained monitoring and assessment program naturally incorporates strategic functions and results in improved criteria, tools, policies, awareness, and legislation. The aggregated database comprises the experience gained by conducting systematic assessments and includes the regular resampling of reference sites and long-term monitoring of reference condition. The database allows comprehensive analysis and interpretation of spatial and temporal trends and tracking the effectiveness of different water quality management programs. The overall program thereby fosters continuous improvement through adaptive management because the relevant information and the interpretation of that information is made available to managers.

As an example, full documentation of the results and benefits of improvements in wastewater treatment on multiple waterbodies in both Ohio and Maine would not have been possible without a comprehensive biological monitoring network and tiered uses to put the results into a communication and management context *(See Case Example 6-4. Long-term Monitoring and Use Re-establishment in Maine).* Furthermore, tiered uses allowed the two States to secure and retain the gains made by upgrading some of the affected rivers to higher tiers, a development that had not been anticipated before the wastewater treatment was improved. These examples also validated the process of setting TALU-based WQS and using them to develop regulatory requirements. The outcomes allayed many of the original uncertainties about the cost-effectiveness of water quality based permitting and gave regulatory programs the confidence to implement new requirements. This was critical in Ohio where the virtues of municipal wastewater treatment more stringent than secondary treatment were widely debated and doubted in the early 1980s. Advanced treatment (also known as best available demonstrated control technology or BADCT) is now widely supported because not only did it work as a treatment technology, but it delivered the end outcome of improved biological condition.

The comprehensive, long-term programs in Ohio and Maine have demonstrated their value by improving prioritization of management actions and enabling more effective targeting of resources. Chapter 6 summarizes several case examples of how biological monitoring and tiered uses contribute to many different aspects of the water quality management cycle (Figure 5-1).

5.4 Key points from Chapter 5

States that have successfully implemented a TALU approach have found that:

1. The specificity of designated uses greatly influences the level of precision at which a water quality management program operates. Incorporating more refined, or tiered, aquatic life uses into water quality standards can have a positive effect on water quality management outcomes. States that have made this transition have demonstrated that tiered aquatic life uses promote both the development of more appropriate aquatic life use goals and biological criteria to measure attainment of those goals.

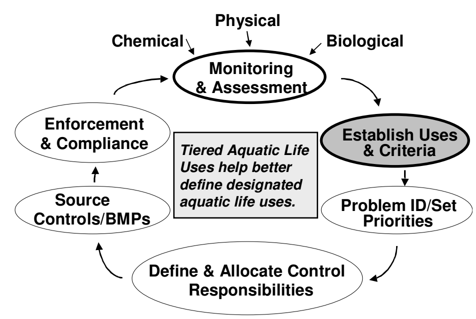
2. Tiered uses in State and Tribal water quality standards, coupled with a systematic and comprehensive monitoring and assessment program, can provide comprehensive support to water quality management programs. In Maine and Ohio, the end result supports baseline CWA management programs such as NPDES permitting, construction grants, and, more recently, the revolving loan program, basin planning (including TMDLs, listings of impaired waters, development of restoration plans), and nonpoint source assessment.

3. Though based on different technical approaches, their development of tiered aquatic life uses followed common tasks and milestones. Development of tiered uses has been most successful when there was early and consistent collaboration among their monitoring, criteria, and standards programs.

CHAPTER 6. HOW HAVE STATES AND TRIBES USED TALUS IN WATER QUALITY STANDARDS AND MANAGEMENT?

Tiered aquatic life uses supported by systematic assessments can provide the information needed for water quality management at watershed, regional, and statewide scales. A comprehensive monitoring and assessment program is a critical aspect of implementation of tiered aquatic life uses. The same data and information that provide baseline status assessments also address watershed-specific management needs such as the appropriate designation of individual waterbodies, TMDL development, and NPDES permits. This chapter presents several case examples in Maine and Ohio of how tiered uses and monitoring contribute to all aspects of the water quality based approach to pollution control (Figure 5-1). These include setting criteria and standards; problem identification and establishing priorities (stressor identification); defining and allocating control responsibilities (source identification); determining source controls or BMPs (TMDLs, UAAs, WLAs); and enforcement and compliance (NPDES permits and other compliance agreements). The following are case examples of how TALUs, coupled with systematic monitoring and assessment, have and can be used to support key water quality management programs and functions. These examples further exemplify what can be accomplished by following the developmental process described in Chapter 5. Accompanying each case example is a diagram of U.S. EPA's Water Quality Management Cycle (Figure 5-1) with the key component for that particular example shaded. Most of the following examples were accomplished during the Program Maintenance phase of the TALU development milestones (Figure 5-2) and demonstrate what can be produced as the bioassessment program matures; however, some of the initial assessments can be accomplished during the Program Implementation phase.

CASE EXAMPLE 6-1. REFINING WATER QUALITY CRITERIA IN OHIO



Ohio EPA developed empirical associations between aquatic life and ambient stressor levels for parameters such as ***dissolved oxygen*** from its monitoring program data beginning in the late 1970s. The known prevalence of organic enrichment from point sources and intensive watershed surveys identified dissolved oxygen (D.O.) as a major stressor limiting aquatic life throughout the 1980s (Ohio EPA 1988, 2000).

When the Exceptional Warmwater Habitat (EWH) aquatic life use was established in 1978, Ohio also established tiered dissolved oxygen criteria to protect "highly sensitive aquatic organisms; growth and reproduction of recreationally and commercially important species; [and] maintenance of populations of imperiled species" (Ohio EPA 1996). This was in contrast to the goal for the Warmwater Habitat (WWH) use, which was the "maintenance of typically representative warmwater aquatic organisms and recreationally important species" (Ohio EPA 1996). The original single criteria for EWH streams of 6 mg/l was largely based on pertinent literature of the time, best professional judgment using the knowledge that these streams supported populations of very sensitive aquatic species, and that the D.O. criteria should be more stringent than the WWH criterion (5 mg/l daily average, 4 mg/l minimum).

Since the original adoption of the EWH use and associated tiered D.O. criteria, analyses of ambient biological and chemical data suggested that the 6 mg/l minimum criterion was over-protective for these waters. Both statewide and reach specific data were used to document streams with dissolved oxygen concentrations below 6 mg/l (but typically above 5 mg/l) that fully attained the EWH aquatic life use as measured by the numeric biocriteria. These results were used to justify a two-number criterion of 6 mg/l average, 5 mg/l minimum for the EWH use (Ohio EPA 1996). Two examples of these data include the stressor-response relationship between grab sample D.O. data (Figure 6-1) and continuous D.O. data (Figure 6-2) and the IBI in the E. Corn Belt Plains (ECBP) and Huron/Erie Lake Plain (HELP) ecoregions of Ohio. Both graphs show an expected gradient of response between D.O. and IBI scores and show that minimum dissolved oxygen values between 5 and 6 mg/l were commonly associated with IBI scores in the EWH range.

Figure 6-1 illustrates a relationship that is commonly observed between stressors and biological measures where multiple stressors are prevalent. On Figure 6-1, to the left of the dashed line at 5.0 mg/l (grab samples), numerous D.O. values are found associated with low IBI scores, but very few at IBI scores above 50 (EWH). If D.O. is >5.0 mg/l, IBI scores are much more likely to attain WWH (>40) and EWH (>50). Figure 6-2 shows continuous D.O. data vs. IBI ranges that correspond to quality tiers ranging from exceptional to very poor. This also supports a similar conclusion as Figure 6-1, but captures the full range of D.O. values that occur over a 24-hour period, especially the early morning hours when the diel cycle yields the lowest values.

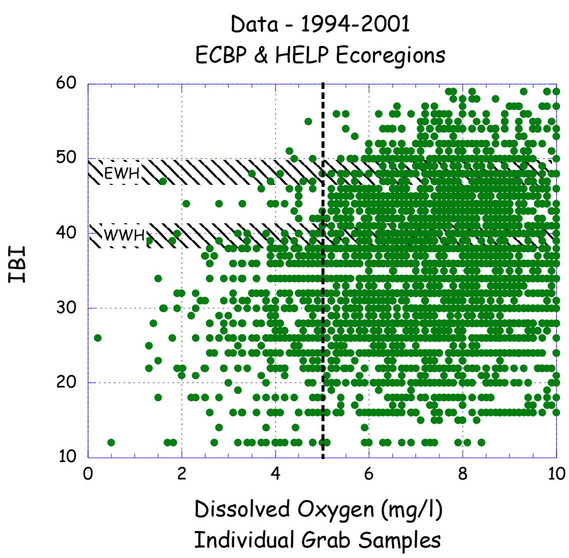


FIGURE 6-1. Dissolved oxygen concentrations (individual grab samples) vs. Index of Biotic Integrity (IBI) values in the Huron/Erie Lake Plain (HELP) and E. Corn Belt Plains (ECBP) ecoregions of Ohio. Hatched areas represent Exceptional Warmwater Habitat (EWH) and Warmwater Habitat (WWH) biocriteria for the ECBP ecoregion.

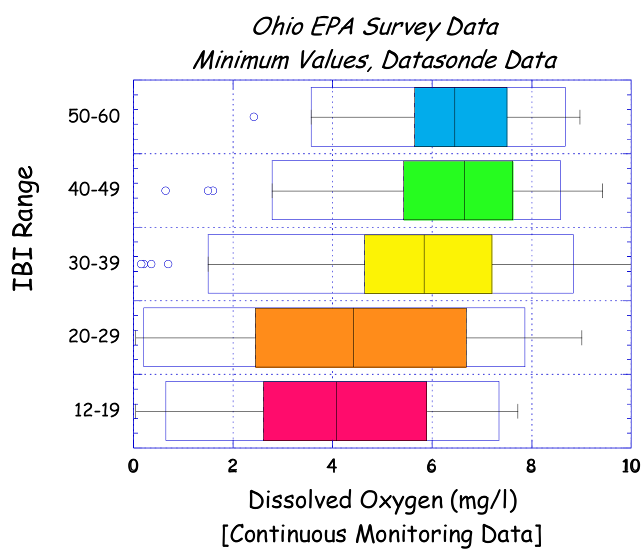
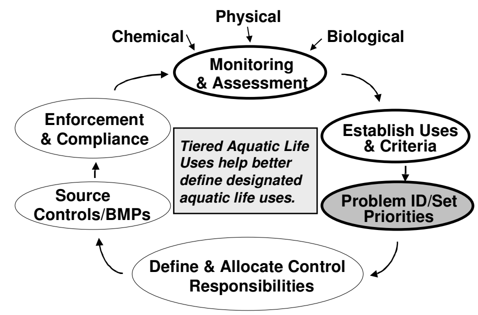


FIGURE 6-2. Box plots of minimum dissolved oxygen concentrations by IBI ranges for continuous monitoring data at all locations monitored in 1988 and 1994. IBI ranges are: 50-60 (exceptional, EWH); 40-49 (good, WWH), 30-39 (fair); 20-29 (poor); 12-19 (very poor).

The key message of this case example is that water quality criteria can be refined to reflect aquatic life use tiers if sufficient ambient data exists over sufficient spatial and temporal scales. It also provides more confidence in applying the water quality criterion as a design target for permitting and TMDL purposes. The previous EWH D.O. criterion (6 mg/l minimum) became a disincentive to redesignate rivers and streams that were fully attaining the EWH biocriteria because of the difficulty in meeting the permit limits. The criterion revision, based in part on the analyses presented here, resolved that situation in the majority of cases and allowed for the redesignation of such rivers and streams to EWH.

CASE EXAMPLE 6-2. DEVELOPMENT OF MORE PRECISE TARGETS FOR RESTORATION IN OHIO



Nutrients have been identified as a major stressor to aquatic life across the U.S. (U.S. EPA 2002b). Nutrients are not directly toxic under most conditions, but rather exert their influence on higher organism groups via interactions within energy pathways and by influencing D.O. dynamics within streams and rivers. Ohio EPA described biological gradients of response to nutrient concentrations in streams and rivers (Ohio EPA 1999a). This was accomplished by linking the primary nutrients (nitrate, total phosphorus) and other parameters to the biocriteria (IBI, ICI, etc.) on a statewide, ecoregion, and stream/river size basis. Thus ranges of these parameters consistent with attainment of the tiered aquatic life uses were accomplished (Ohio EPA 1999a; Table 6-1). While the values in Table 6-1 are not explicit water quality criteria, they are used as TMDL targets given the direct linkage they have with aquatic life use attainment. In addition to ambient fish and invertebrate data, ambient chemical data, and stream habitat data, Ohio is currently collecting information on chlorophyll and algal assemblages to improve understanding of the mechanisms of nutrient impact on aquatic life (Bob Miltner, Ohio EPA, personnel communication). This work should result in refined targets that can be used to determine which restoration activities should be most effective at restoring aquatic life. The identification of nutrient targets for each aquatic life use tier provides an appropriate and achievable level of protection for specific waterbodies. This application provides restoration targets for TMDLs that, if achieved, should result in full attainment of aquatic life uses.

TABLE 6-1. Statewide total phosphorus targets (mg/L) for Ohio rivers and streams.

Watershed Size	Aquatic Life Use		
	EWH	WWH	MWH
Headwaters (drainage area <20 mi^2)	0.05	0.08	0.34
Wadeable rivers (20 mi 2 <drainage area <200 mi	0.05	0.10	0.28
Small rivers (200 mi 2 <drainage area <1,000 mi	0.10	0.17	0.25
Large rivers (drainage area >1,000 mi	0.15	0.30	0.32
EWH =Exceptional Warmwater Habitat; WWH =Warmwater Habitat; MWH =Modified Warmwater Habitat			

As for nutrients, Ohio does not have explicit ***habitat and sediment*** criteria in the WQS. However, targets for habitat and sedimentation outcomes were developed by demonstrating a relationship between specific good quality and poor quality attributes and their ratios. Unlike water quality parameters, single numeric criteria for habitat and sedimentation do not exist and are inappropriate because 1) there are complexities in identifying expected values or ranges of values for specific attributes, 2) the resultant effects on the aquatic biota are explained by aggregations of good (warmwater) and poor (modified; see HIMA in Table 6-2) habitat attributes, and 3) the spatial scale over which these stressors exert their effects on aquatic life includes multiple dimensions (Rankin 1995). Rather than generating tiered criteria for habitat and sediment attributes, Ohio has developed quantitative habitat and sediment targets for TMDLs based on regional stream types (e.g., low vs. high gradient) and stream-size dependent “dose-response” relationships with the numeric biocriteria associated with the tiered aquatic life uses (Rankin 1995). The Stillwater River TMDL (Ohio EPA 2004) in the E. Corn Belt Plains (ECBP) ecoregion is an example of how nutrient, sediment, and habitat targets (“criteria”) were developed and used along with more traditional chemical criteria to direct TMDL development in the watershed (Table 6-2).

TABLE 6-2. Numeric targets for biological, habitat, and water quality parameters for the Stillwater River in western Ohio. From Ohio EPA (2004) TMDL report for the Stillwater River watershed. The targets and criteria vary in accordance with the tiered uses, which are resolved prior to impaired water delineations and TMDL development.

	Biological Criteria		Habitat Targets		Water Quality Criteria				Nutrient Targets		
					Ammonia-N*		Dissolved Oxygen*				
Aq. Life Use	Min. ICI	Min. IBI	QHEI	HIMA[a]	Max	Mean	Min	Mean	TKN[b]	Nitrate[b]	TP[b]
MWH	22	24	45	≤3	7.3	1.2	3.0	4.0	4.0	3.0	0.30
WWH	32	36	60	≤1	7.3	0.8	4.0	5.0	1.0	1.0	0.08
EWH	42	46	75	0	4.5	0.8	5.0	6.0	1.0	0.5	0.05

[a]HIMA - High Influence Modified Habitat Attributes
[b]Target values are adopted from Ohio EPA (1999)
*Specific numeric water quality exist in OAC 3745- 1-07, Tables 7-3 through 7-8; target values are guidelines based on the 75[th] percentile values of temperature (24°C) and field pH (8.1) from all samples collected during the 1999 Stillwater survey.
MWH = Modified Warmwater Habitat; WWH = Warmwater Habitat; EWH = Exceptional Warmwater Habitat

All of the targets in Table 6-2 were either wholly or partially generated based on responses between the parameters, biological assemblage data, and the tiered aquatic uses to which they are related. This is important because most of these parameters, habitat in particular, are not amenable to the traditional laboratory based derivation. When these parameters are altered from "naturally occurring" conditions, they can induce an adverse response for the biota, thus behaving as stressors. Targets for TMDLs or other restoration strategies would either be difficult to generate, or lead to potentially incomplete solutions without being ground-truthed in ambient data relationships and a tiered aquatic life use framework, the latter of which is typically associated with a stressor gradient based on habitat or landscape characteristics. Since many of the targets in Table 6-2 were generated directly from ambient stressor and response relationships, their interpretations are likely less ambiguous than a rote application of lab derived criteria, although causative associations may be weaker. This approach is consistent with a recommendation in the NRC TMDL report (NRC 2001) that criteria or targets be positioned as closely as possible to the designated use and that indicators representing the full causal chain of events from stress to exposure to response be used.

Understanding the role of habitat as an influence on the biological restoration potential for a waterbody may be one of the greatest values of tiered aquatic life uses coupled with a systematic assessment process. Habitat and landscape changes compose a common stressor gradient along which States and Tribes may derive tiered uses. Tiered uses provide a useful framework for evaluating restoration potential, prioritizing management actions, and allocating abatement resources.

CASE EXAMPLE 6-3. DETERMINING APPROPRIATE LEVELS OF PROTECTION IN OHIO

Hurford Run is a small stream located in an ***urban/industrial*** area (steel finishing, petroleum refineries) of Canton, Ohio that drains an area of 8.5 square miles (Figures 6-3, 6-4). The entire stream has been subjected to direct channel modifications from the 1900s up to the time of the study. During the biological surveys in the mid 1980s, the stream was severely impaired by chemical pollutants, so much so that some sites had no fish. Because of the severity of the impairment, the use attainability analysis (UAA) relied on the assessment of habitat quality by the Qualitative Habitat Evaluation Index (QHEI; Rankin 1995).

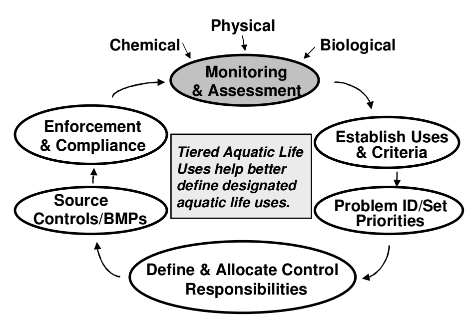


FIGURE 6-3. 1986 photograph of Hurford Run near Canton, Ohio looking upstream at the reach that is classified as a Limited Resource Water. Disturbed soil was caused by efforts to remove soils contaminated by nearby industrial operations.

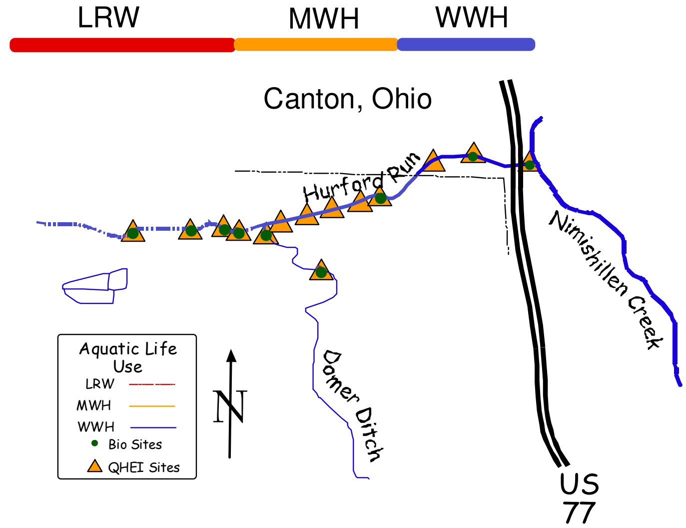


FIGURE 6-4. Map of Hurford Run near Canton, Ohio showing Ohio EPA IBI (solid circles) and habitat (QHEI, triangles) sampling stations. Spatial extent of stream aquatic life use designations is denoted along the top.

Established relationships between attributes of habitat as measured by the QHEI and levels of biological performance consistent with the tiered aquatic life uses provide an important tool to evaluate use attainability and assign appropriate uses to specific streams and rivers (Rankin 1989, 1995; Ohio EPA 1990). For example, Ohio has identified which habitat features may limit aquatic communities and which are predictive of streams with warmwater (WWH) and exceptional warmwater (EWH) biological communities. Figure 6-5 summarizes the IBI (left) and QHEI scores (right) for Hurford Run from 1985 to 1998. Very poor habitat quality from recent and historical channelization in the upper reach (RM 1.8 - 2.5) of Hurford Run and the associated hydrological characteristics (e.g., ephemeral flows) resulted in a Limited Resource Waters (LRW) designation for this upper reach. The middle reach beginning at the confluence of Domer Ditch (RM 1.7-1.0) was subject to extensive, maintained channel modifications and resulted in degraded habitat features (Figure 6-5, right), but water was always present. Channel maintenance practices resulting in poor quality substrates, undeveloped pools and riffles, and a lack of instream cover preclude biological recovery to assemblages consistent with the WWH use. Following a use attainability analysis (UAA), the middle reach was designated as Modified Warmwater Habitat (MWH), reflecting the biological restoration potential for a channel-modified stream.

The lower one mile of Hurford Run, although previously relocated and channelized, naturally recovered sufficient warmwater (good) habitat attributes such as coarse substrates and better developed riffle and pool features to achieve QHEI scores (>60-70) that are typical of the WWH use for this ecoregion, hence this segment was left at WWH. The tiered aquatic life uses that were assigned represent the highest attainable potentials given the existing level of sanctioned channel maintenance in this urban stream.

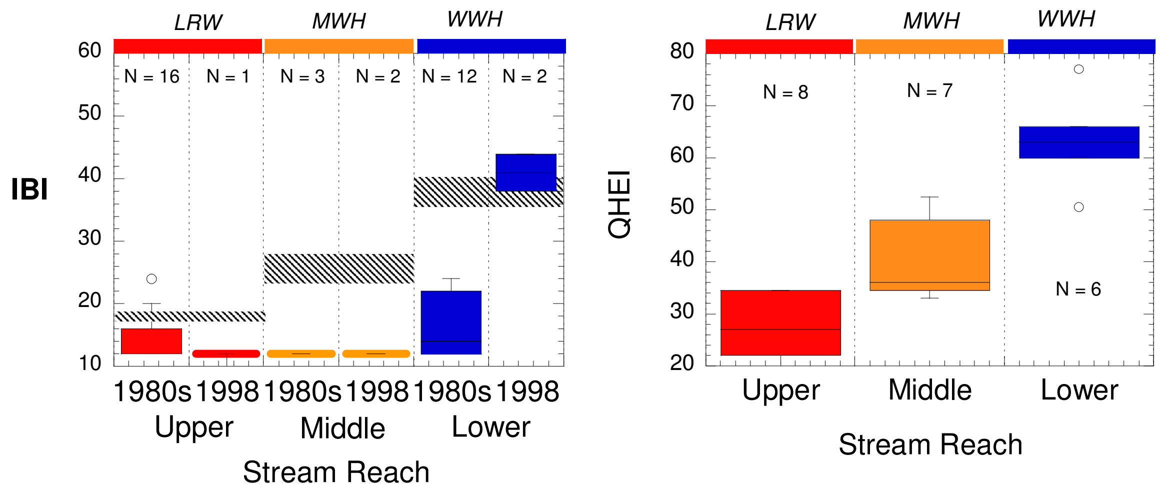
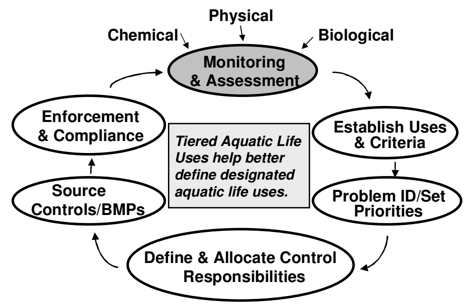


FIGURE 6-5. Box and whisker plots of IBI (left) and QHEI (right) by stream segment in Hurford Run near Canton, Ohio. Aquatic life use designations for segments are denoted along the top of each plot. 1998 data is separated from the 1980s data for the IBI, but data are combined for the QHEI. Data collected between 1985 and 1998. Lines are sites with no variability in scores (IBIs = 12). The hatched bars denote Ohio biocriteria for each tiered use.

All of the designated uses required additional abatement of the major point sources discharging to Hurford Run. Following the initial abatement of point source discharges in the late 1980s, data collected in 1998 demonstrated recovery of the IBI score near the mouth of the stream to the WWH biocriterion as predicted by the QHEI (Figure 6-5, left). Because this reach was designated WWH, it is protected from any further alteration below this quality. The MWH designated middle reach and LRW designated upper reach of Hurford Run have been subjected to ongoing channel maintenance activities (e.g., dredging, bank mowing), which has limited the amount of biological restoration that can be expected. However, even these less-than-CWA goal uses are impaired due to unresolved toxic impacts (reflected in very poor IBI scores; Figure 6-5, left) presumably from the point sources and/or legacy impacts associated with the industrial sites bordering the stream.

Urban/industrial streams such as Hurford Run present challenges in terms of setting and attaining restoration goals. Visually, the lower reach of Hurford Run may not exemplify the classic depiction of a natural stream because of its urban/industrial setting and location adjacent to major highways. The instream habitat, however, indicated a WWH potential, which was eventually verified as the effects of chemical stressors were reduced. The feedback provided by bioassessments based on the systematic collection of biological and habitat data, which is essential to using tiered aquatic life uses, is an important impetus for achieving water quality goals.

CASE EXAMPLE 6-4. LONG-TERM MONITORING AND USE RE-ESTABLISHMENT IN MAINE



Between 1974 and 1981, an estimated 33 million dollars was spent by industry, State, and federal sources to implement primary and secondary wastewater treatment technology on facilities discharging into a 100 km section of the Penobscot River between Millinocket and Costigan, Maine. These expenditures resulted in an 80% reduction in loadings of biochemical oxygen demand and total suspended solids discharged from the kraft and sulfite pulp and paper mills in the study area. In 1974, the benthic macroinvertebrate community was determined to be highly degraded at three stations in closest proximity to pulp and paper effluents (Stas. 129, 131, 133). An additional two sites, somewhat downstream of pollution outfalls (Stas. 125, 126), were determined to be degraded (Rabeni 1977). The benthic community of the study area has been re-evaluated several times following major water quality changes in the 1970s, with the conclusion that the investments have resulted in dramatic improvements in the river's ability to support aquatic life.

Station 129 is located 4 km downstream of the Lincoln Pulp and Paper Company outfall. Figure 6-6 provides a graphical summary of changes in two metrics of aquatic community structure for the period of record at Station 129. Maine DEP uses the metrics shown in a linear discriminant model to assign aquatic life classification attainment. In 1974, Station 129 was designated as "highly polluted." The substrate at Station 129 was covered with sewage bacteria (*Sphaerotilus*) and the invertebrate community was restricted to worms, leeches, and pollution tolerant midge larvae. Numbers of individuals were very high, indicating a "bloom" of tolerant, opportunist organisms. Diversity and richness values were very low (Figure 6-6), and there was a complete absence of pollution-sensitive mayflies and stoneflies. In terms of aquatic life classification, this station did not meet minimum State or federal standards.

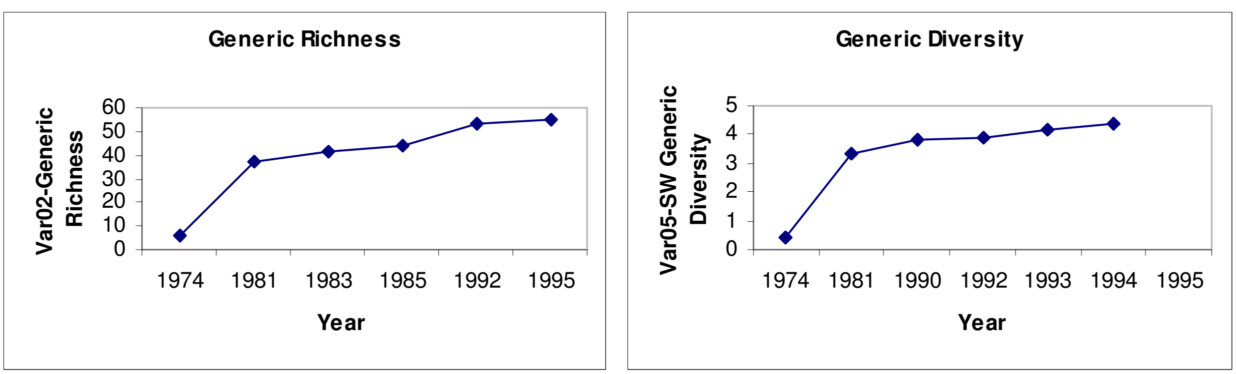


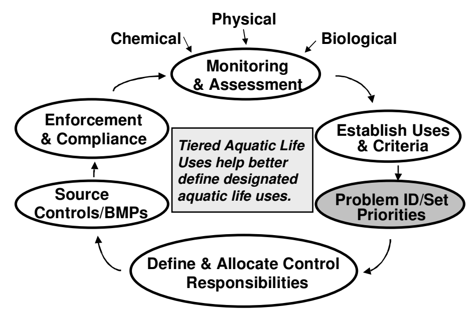
FIGURE 6-6. Scatter plots showing values for two biological community variables, generic richness (left) and generic diversity (right), from Sta. 129, the Penobscot River below Lincoln Pulp and Paper, between 1974 and 1996.

Dramatic improvements in the benthic macroinvertebrate community were evident by 1981 (Davies 1987). Total abundance was down, richness and diversity were greatly improved (Figure 6-6), and the proportion of tolerant midge larvae was lower. Low numbers of stoneflies and mayflies were also present. Overall, attainment had improved to Class C standards. The station has been sampled four times since 1981, each time meeting Class B standards and showing continued improvement in community structure, including high diversity and richness and healthy stonefly and mayfly populations. This long-term dataset provides a valuable example of the responsiveness of biota to water quality improvements. It also highlights the unique usefulness of biological monitoring to document and summarize the real world benefits of responsible stewardship of aquatic resources.

As a result of investment in wastewater control, the Penobscot River improved dramatically, from not attaining Class C standards in 1974 to attaining Class B standards throughout most of the river today. As a result, Maine upgraded the river from Class C to Class B in two steps. As of 1999, the entire mainstem, with the exception of an impounded section, is now Class B and must attain Class B standards. Without TALUs, the upgrade could not have taken place and the river would be maintained today as the equivalent of Class C. With Maine's TALUs, the river is now protected as Class B, which has been demonstrated to be attainable throughout. Documentation of the improvement and subsequent protection of the improved conditions is not possible without TALUs.

In addition to the Penobscot, many other streams in Maine have been upgraded in class as a result of effective wastewater treatment or dam removal, which has led to dramatic improvements in biological condition and class attainment.

CASE EXAMPLE 6-5. DEVELOPMENT OF LIMITS FOR NPDES PERMITS IN MAINE



Decoster Egg Farm, located in Turner, Maine, is the largest producer of brown eggs in New England. The Farm has a long history of environmental concerns including levels of ammonia and nitrates in violation of drinking water standards. This case example presents a unique example of the ***detection of biological impacts*** in a stream attaining surface water quality standards but affected by polluted groundwater recharge. Permitting staff had recorded nutrient levels in leachate draining poorly managed manure and chicken carcass waste piles. Stream violations were not sufficiently high to trigger enforcement action based on surface water quality violations but the high levels resulted in contaminated leachate entering groundwater on the Decoster property. In 1989, the Department brought enforcement action against Decoster Egg Farm to prohibit any further spreading of manure on the property and to enforce proper management of other animal waste products.

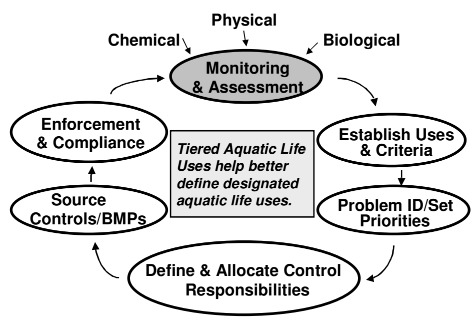
In 1991, the company was required to evaluate the condition of the aquatic life in streams affected by leachate or groundwater upwelling. Two of the streams, Lively Brook and House Brook, were designated by the State to maintain Class B water quality conditions. The use designation process had deemed this to be an appropriate management goal for these streams based on the tiered use designations of other streams of comparable habitat and watershed condition. Field investigations included probes of the hyporheic zone (the water flowing through the stream substrate) to measure the conductivity of the upwelling groundwater. Conductivity is a measure of the ionic strength of water and is a very good means of detecting certain types of pollutants. The streambed investigation uncovered several areas of contaminated groundwater recharge to the stream. Aquatic life sampling, completed in 1992, confirmed impacts to the benthos at three stations affected by groundwater upwelling on Lively Brook and one station on House Brook. Station 188, on House Brook, is located downstream of a failing treatment system that receives waste from the egg washing operation. The waste stream is severely contaminated by nitrates. This station failed to attain minimum Class C aquatic life standards in 1992. Repeat sampling in 1997 demonstrated attainment of Class C standards but the stream still failed to attain its assigned Class B status, indicating the need for additional management intervention. Biomonitoring information was used to issue a consent order requiring termination of manure spreading practices and improved treatment of the products of the egg washing facilities. The egg washing facility was removed.

The Lively and House Brooks case study illustrates the full water program cycle (Figure 5-1). Monitoring and characterization of the habitat and watersheds of the two streams revealed that, with best management practices in place, they should be able to attain Class B status, but in fact were not attaining minimum Class C status. Problem identification showed that contaminated groundwater due to poor management practices was causing the impairment. A set of source controls were applied, the facility complied with the controls, and monitoring of the streams' condition continued. The monitoring showed that although the streams had improved to Class C, they were still not attaining their designated Class B status. Maine DEP applied further source controls on the facility to achieve Class B status.

Ongoing monitoring, iterative management intervention, and tiered use goals confirmed that the streams had the potential to attain Class B status. Without tiered uses, source controls would have stopped when a minimal condition was reached (consistent with a Class C condition) and the two streams would never have recovered to Class B. Tiered aquatic life uses create attainable goals and best uses for waterbodies, resulting in better quality waters than are possible with a single use. If a general aquatic life use system had been in force, it likely would have resulted in a biological quality comparable to Maine's Class C, with no impetus for improvement to the actual potential (Class B).

CASE EXAMPLE 6-6. NPDES PERMITTING AND USE ATTAINABILITY ANALYSIS IN OHIO

Ecologically-based TALUs, a systematic approach to monitoring and assessment, and a sound UAA process can provide substantial benefits for NPDES permitting related to both the derivation of permits and assessing the effectiveness of a permit in restoring an aquatic life use. A system for identification of the attainable potential for the aquatic life of a waterbody using a systematic approach can set credible restoration goals and support measured responses to environmental risks. This case example illustrates the use of TALUs, systematic monitoring and assessment, and a consistent process for conducting UAAs in support of NPDES permitting issues.



The Ottawa River in northwest Ohio has been heavily polluted for more than a century. The river is impacted by the city of Lima, rural communities, and agricultural activities (row crops). Heavy industry in Lima was identified as a major source of water pollution since the 1880s (Leeson 1885 c.f. Ohio EPA 1992) being especially severe in the 1960s " . . . when more than 37 miles were devoid of fish, including the Auglaize River downstream from the Ottawa River" (Ohio EPA 1992). Point sources include one major municipal and two major industrial discharges, industrial contributors to the Lima sewer system, combined sewer overflows (CSOs), and partial or untreated sewage discharges from semi-rural areas in the watershed. The effluent flow from the three major point sources enter the Ottawa River within a 0.8 mile reach and comprise the majority of the river flow during dry weather months. Improvements consistent with CWA technology standards have been made at the major wastewater treatment facilities since the late 1970s. The major causes of impairment include organic enrichment and low D.O., general toxicity, habitat alterations (impoundments), nutrients, ammonia, heavy metals, oil and grease, and chlorine in both the water column and bottom sediments (Ohio EPA 1998).

This case example focuses on a 25-mile segment of the Ottawa River that is directly impacted by major point sources (Figure 6-7) and includes zones of immediate and acute impacts and various phases of recovery downstream. Physical habitat in the mainstem downstream from the major point sources is good to excellent, and the mainstem is designated WWH as the result of a use attainability analysis and upgrade conducted in the late 1980s. Prior to this analysis, most of the river was assigned the Limited Warmwater Habitat (LWH) aquatic life use, which was assigned to rivers thought to be so polluted that restoration was considered unfeasible. The LWH use was developed and applied prior to the development and adoption of TALUs by Ohio EPA and is no longer used.

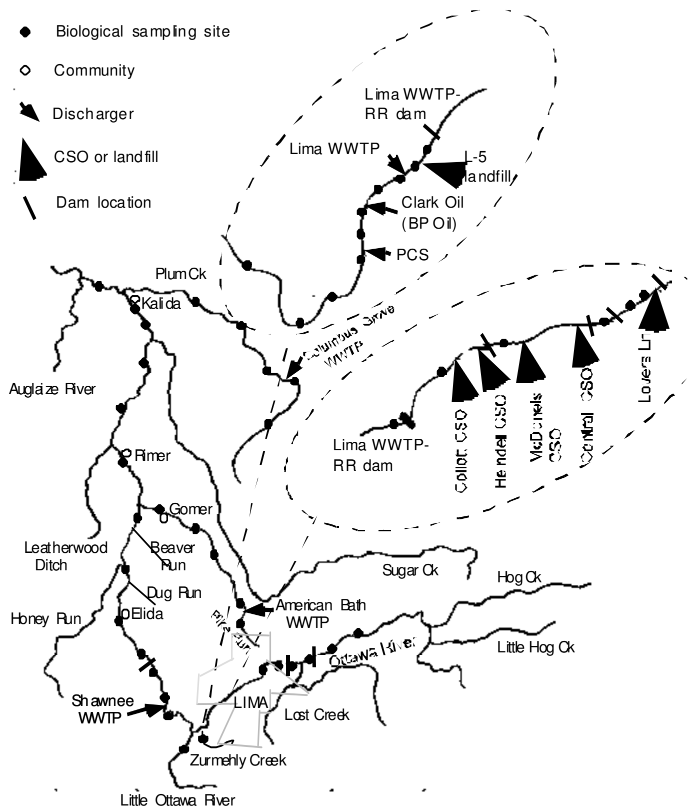


FIGURE 6-7. Map of the Ottawa River with magnification of two reaches in the Lima, Ohio area (after Ohio EPA 1998).

Toxic stressors, exposures, and responses reached a maximum in the segment directly impacted by the three major point sources (Ohio EPA 1998; Yoder and DeShon 2003). Evidence of multiple toxic exposures occurred in the water column chemistry, sediment chemistry, whole effluent toxicity, frequency of DELT anomalies, fish tissue contaminants, and biochemical markers (Table 6-3). These indicators pointed strongly to impacts of a toxic character and the biological response signatures provided the corroborating feedback. Low D.O. can occur in the Ottawa River (Ohio EPA 1998), but the more serious toxic effects that are evident in the biological response signatures presently mask its less serious effects.

TABLE 6-3. A matrix of stressor, exposure, and response indicators for the Ottawa River mainstem based on data collected in 1996 (after Ohio EPA 1998). The darkness of shading indicates the degree of severity of effect or exceedance expressed by an indicator.

SEGMENT	DES. USE	RESPONSE INDICATORS				EXPOSURE INDICATORS						STRESSORS				
	Attain-ment Status	QHEI	IBI	MIwb	ICI	Water Chem.	Sedi-ment Chem.	Tox-icity	% DELT	Fish Tiss.	Bio-marker	# Dams/ Pools	Urban-Indust. Landuse	Cumulative Loads	Spills	CSO SSOs
Ottawa River mainstem - 1996																
Thayer Rd to Sugar St.	FULL-PART.	68	Fair-Good	Fair-Good	Good	Nitrates	Low	NA	Mod-High	Mer-cury	Low	Mod-e	Low	Low	Low	Low
Sugar St. to Lima WWTP	**NON**	47	**Poor** to Fair	**Poor** to Fair	**Poor** to M.G.	CBOD TSS D.O.	As,Cr Cd,Cu Ni,Zn	Mod-erate	**High**	Pesti-cides	BUN Naph B(a)p	**High**	**High**	Mod-erate	Mod-e	**High**
Lima WWTP Allentown dam	**NON**	72	**Poor**	**Poor** to Fair	Fair to Good	Amm. CBOD TSS D.O. Nitrates Phos Chrom. PAH Pesticid	As,Cr Cd,Cu Ni,Zn PAH	Mod-erate	**Very High**	Selen-ium Pest-icides	EROD Naph B(a)p BUN	Mod-e	**High**	**High**	**High**	**High**
Allentown dam to Kalida	PAR-TIAL	69	**Poor** -Fair	Fair-Good	Good -Exc.	TSS	Low	NA	**High**	Pesti-cides	Low	Low	Low	**High**	Low	Low
Kalida to mouth	FULL	69	Good	Good	Exc.	TSS	Low	NA	**Very High**	Pesti-cides	Low	Low	Low	**High**	Low	Low

QHEI scores for the Ottawa indicated more than adequate habitat to support the WWH use designation (Rankin 1989, 1995). In a growing recovery zone immediately below the impacted reach, the biota eventually exhibited recovery to WWH status in the lower reaches of the river. In the impaired sections, the biological response signatures strongly indicate general toxicity, which is a fundamentally different response than what would occur in response to habitat or low D.O. alone (Figure 6-8; Yoder and Rankin 1995b; Yoder and DeShon 2003). Results from a similar time period for the Scioto River are shown for comparison. This river is impacted by non-toxic causes and sources including organic enrichment and oxygen demanding wastes from sources that dominate the low flow of the river and emanate from a similar municipal infrastructure and watershed setting. Taken together, these considerations led Ohio EPA to redesignate (upgrade) the Ottawa River from LWH to WWH in 1989. The redesignation was controversial and resulted in legal actions challenging the WWH use. Plaintiffs contended that the habitat could not support a WWH assemblage and further argued that D.O. concentrations consistent with WWH criteria were unattainable due to upstream impoundments and the flow regime. The WWH designation was upheld because Ohio had a substantial record demonstrating the relationship between habitat condition (as QHEI) and attainable biological condition described in the tiered uses. The response signatures indicated that the cause of non-attainment in the Ottawa River was primarily toxicity.

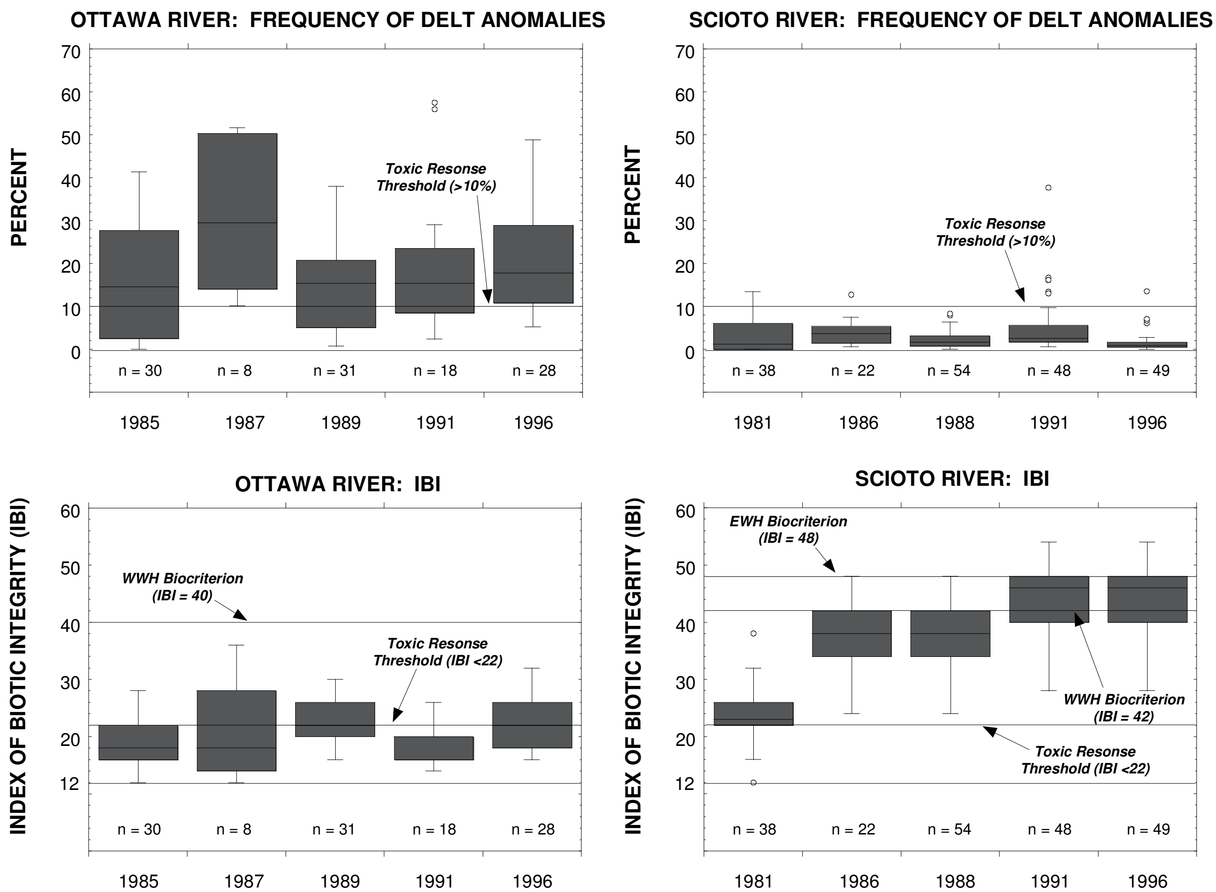


FIGURE 6-8. Results for two key fish assemblage measures (%DELT anomalies, upper left panel and IBI, lower left panel) showing the thresholds for toxic responses in the Ottawa River study area between 1985 and 1996. The results are shown with those from the Scioto River between 1981 and 1996 to illustrate the different responses shown in a river impacted by non-toxic stressors.

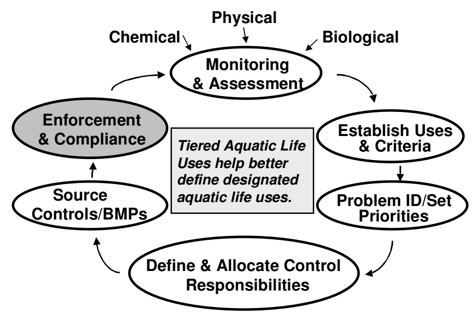
The WWH redesignation and the subsequent permitting of the three major point sources could have taken a significantly different path in the absence of the TALU approach employed by Ohio EPA. Instead of keeping the focus on the most limiting problem of complex toxicity, the outcome could have been diverted by the initial claims of habitat limitations and D.O. issues. Ohio's systematic approach to monitoring directly tied to its TALUs was upheld in a court case on the redesignation to WWH, which has averted subsequent legal actions in other similar permitting cases. This is related to the soundness and consistency of the UAA approach and the perception that the TALUs are reasonably attainable and protective.

One tool the NPDES program uses to identify potential problems from dischargers is non-compliance with permit terms and conditions. In this case, none of the individual point sources involved were considered in non-compliance of their NPDES permits at the time of the assessments. However, their cumulative effect on biological condition resulted in severe biological impairment of the river. As a result, Ohio EPA imposed controls to significantly improve water quality, including chronic WET limits, close scrutiny of intermittent releases and spills, and internal audits conducted by two of the industrial facilities involved. In addition, an unregulated landfill leachate was discovered and subsequently required remediation.

Under a tiered system, the biocriteria endpoints vary with the specific use and thus can affect the NPDES permit. For example, a WWH designation requires better biological condition (higher IBI, MCI and MIwB scores) than the LRW use. Accordingly, LRW waters can tolerate higher nutrients and lower D.O. than WWH waters (See Figure 6-2, Table 6-2, and Appendix B), which would affect permit limits. A decision that the stream was either habitat limited or dissolved oxygen limited alone would have diverted attention away from the severe toxic impacts that were in reality limiting the aquatic life in this river. The magnitude of these influences would have been underestimated on the sole basis of administrative measures, without the stressor analysis that identified the causes of impairment in the Ottawa River.

CASE EXAMPLE 6-7. SUPPORT FOR DREDGE AND FILL PERMITTING IN OHIO

The losses of habitat diversity or habitat-mediated stressors such as increased siltation are now the most prevalent causes of aquatic impairment in Ohio (Figure 6-9, Ohio EPA 2000). This is also true across much of the U.S. (U.S. EPA 2002b). Environmental effects of extensive landscape changes and in stream habitat alterations are a primary stressor gradient along which the tiered aquatic life uses were developed. Some habitat alterations are readily restorable while others are essentially permanent either because they are continuously maintained for flood control or drainage purposes or they exceed the natural capacity for recovery.



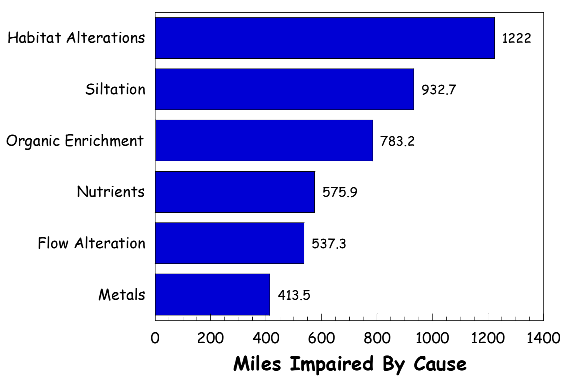


FIGURE 6-9. Six leading causes of aquatic life impairment in Ohio up to the year 2000 (from Ohio EPA 2000).

States can use Sections 401 and 404 of the CWA to manage direct alterations to aquatic habitats. Tiered aquatic life uses have proved useful in 404 permitting and 401 certification of those permits. Those wanting to modify a stream that will result in the discharge of dredge or fill material into waters of the U.S. must obtain a Section 404 permit from the U.S. Army Corps of Engineers (ACOE) and a Section 401 water quality certification from the State. The State must certify that proposed activities will comply with, not violate, WQS. The existence of biocriteria in the Ohio WQS makes this linkage a valid tool for evaluating the impacts of habitat alterations that are covered under the CWA. Ohio EPA used a 20+ year database to develop habitat stressor gradients along several aspects of habitat quality at both site and watershed scales, including overall habitat quality as measured by the QHEI and for specific attributes such as substrate and channel condition. Examples of these stressor gradients from the E. Corn Belt Plains (ECBP) and Huron/Erie Lake Plain (HELP) ecoregions are illustrated in Figure 6-10.

Tiered aquatic life uses have enabled a range of management responses to dredge and fill projects related to the quality and sensitivity of the waterbody in question. Tiered uses are an important consideration in the implementation of nationwide permits. Nationwide permits are designed to minimize site-specific oversight where ecological risks are assumed to be low. Frequently, however, the criteria for which places are eligible can overlook high quality waters and lead to their alteration. The Ohio EWH use designation requires high habitat quality and stable hydrological regimes (especially in headwater and wadeable streams). Because these essential attributes can be altered by direct modifications to the stream channel and other habitat features, Ohio requires individual reviews of projects that occur in such high quality streams. Under a general use system, these would be lumped with all other streams under the nationwide permit system.

The same information embodied in the tiered aquatic life uses allows Ohio to expend less oversight on streams that cannot attain the WWH use designation. Such streams are generally ephemeral or continuously maintained as drainage conveyances. This does not mean that physically degraded streams are ignored. The attention gained by habitat impacts has prompted the development of mitigation standards that will take the tiered aquatic life uses into account and require enhancement or restoration wherever feasible. The stressor-response relationships (Figure 6-10) that have been developed between biological assemblages and key habitat attributes have been applied to the 401 program in Ohio. For nationwide 404 permits a series of general and specific exclusions and conditions have been derived that vary with tiered aquatic life uses (ACOE 2002). These include a general exclusion (of nationwide permits) for streams that are EWH and for certain antidegradation tiers (State Resource Waters and Outstanding State Resource Waters), the delineation of which was based primarily on the same biological assemblage attributes that are in common with Ohio's tiered aquatic life uses.

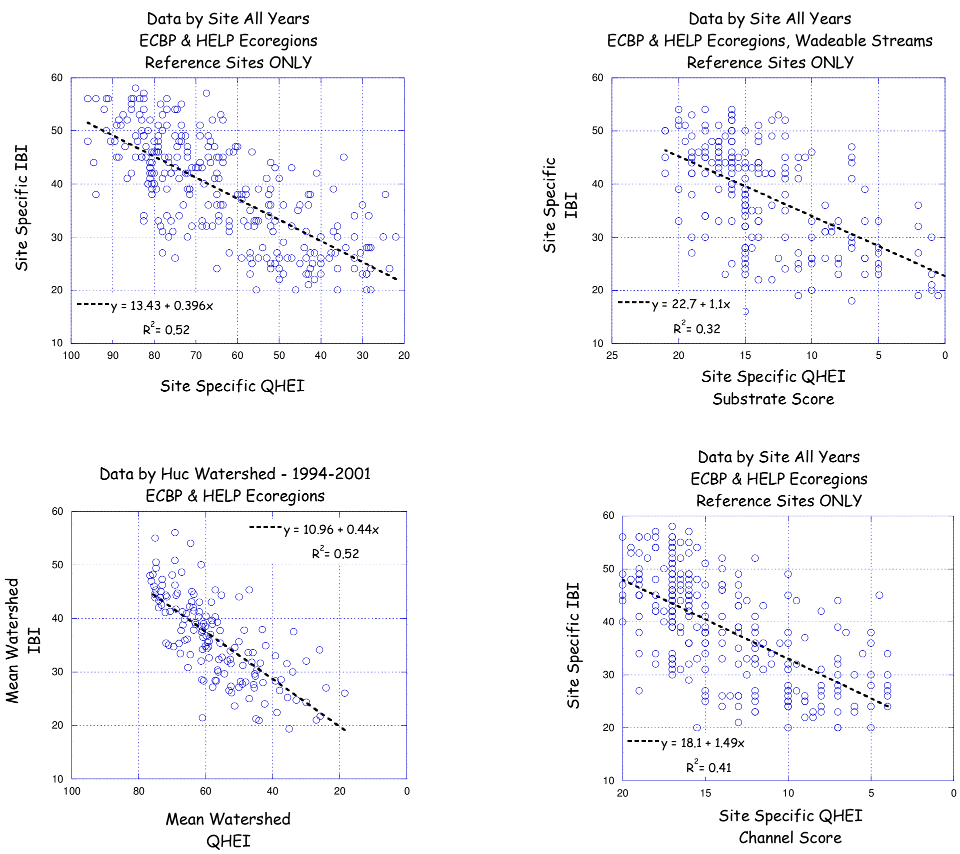


FIGURE 6-10. Examples of habitat stressor gradients vs. IBI for Ohio wadeable streams in the ECBP and HELP ecoregions.

Aside from the general considerations discussed above, tiered uses have also proved useful for specific nationwide permits. For example, Nationwide Permit 21 is for surface coal mining activities. Higher quality uses such as WWH or EWH and Coldwater Habitat (CWH) require individual 404 permits in all cases. Only MWH or LRW uses can be exempted from site-specific review under a nationwide permit for mining (and for these there are stream length limitations). Again this is a significant benefit of having tiered uses and the knowledge of the relationships between activities (e.g., habitat alterations) and the biological responses in the indexes that compose the tiered biocriteria. The 404/401 program in Ohio is still evolving. One goal is to move away from a case-by-case review of every permit by developing mitigation standards tied directly to the tiered aquatic life uses that will be protective, relatively rapid, accurate, and efficient in terms of resource expenditures. Making similar decisions within a single use system would be more difficult and require either more case-by-case oversight to account for habitat gradients, or risk being over-protective in some cases and under-protective in others.

References & Additional Resources

Aarts, B.G.W., F.W.B. van den Brink and P.H. Nienhuis. 2004. Habitat loss as the main cause of the stagnating recovery of the fish faunas of regulated large rivers in Europe: the transversal floodplain gradient. *River Research and Applications*.

Adams, S.M. 1990. Status and use of biological indicators for evaluating the effect of stress on fish. *American Fisheries Society Symposium* 8(1).

Adams, S.M., K.D. Ham, and R.F. LeHew. 1998. A framework for evaluating organism responses to multiple stressors: mechanisms of effect and importance of modifying ecological factors. In *Multiple Stresses in Ecosystems*, J.J. Cech, B.W. Wilson, and D.G. Crosby (eds.), pp.13-22. Lewis Publishers, Boca Raton, FL.

Allan, J.D. 1995. *Stream Ecology: Structure and Function of Running Waters*. Chapman & Hall, New York.

Allan, J.D., D.L. Erickson, and J. Fay. 1997. The influence of catchment land use on stream integrity across multiple scales. *Freshwater Biology* 37:149-161.

Allan, J.D. 2004. Landscapes and riverscapes: The influence of land use on stream ecosystems. *Annual Review of Ecology, Evolution, and Systematics* 35:257-84.

Allen, A.P. and R.J. O'Connor. 2000. Interactive effects of land use and other factors on regional bird distributions. *Journal of Biogeography* 27:889-900.

Angermeier, P.L. and J.R. Karr. 1994. Biological integrity versus biological diversity as policy directives. *BioScience* 44:690-697.

Appelberg, M., B.I. Henrikson, L. Henrikson, and M. Svedang. 1993. Biotic interactions within the littoral community of Swedish forest lakes during acidification. *Ambio* 22:290-297.

Armitage, P.D., D. Moss, J.F. Wright, and M.T. Furse. 1983. The performance of a new biological water quality score system based on macroinvertebrates over a wide range of unpolluted running-water sites. *Water Research* 17:333-347.

Army Corps of Engineers (ACOE). 2002. Memo: Grant of Section 401 Certification Authorization of discharges of dredged or fill material to various waters of the state [Ohio] for Nationwide Permits as detailed in the January 15, 2002, Federal Register (Volume 67, Number 10). CECW-OR Washington, D.C. 20314-1000.

Bailey, R.C., R.H. Norris, and T.B. Reynoldson. 2004. *Bioassessment of Freshwater Ecosystems Using the Reference Condition Approach*. Kluwer, Dordrecht, NL.

Baird, J.D., and G.A. Burton, Jr. (eds.). 2001. *Ecological Variability: Separating Natural from Anthropogenic Causes of Ecosystem Impairment*. SETAC Press, Pensacola, FL. 336 pp.

Bakus, G.J., W.G. Stillwell, S.M. Lather and M.C. Wallerstein. 1982. Decision-making: with application for environmental management. *Environmental Management* 6:493-504.

Ballentine, L.K. and L.J. Guarraia (eds.). 1977. *Integrity of Water*. EPA 055-001-010-01068-1. U.S. EPA, Office of Water and Hazardous Materials, Washington, DC.

Barbour, M.T., J.M. Diamond, and C.O. Yoder. 1996a. Biological Assessment Strategies: Applications and Limitations. In *Whole effluent toxicity testing: An evaluation of methods and prediction of receiving system impacts*, D.R. Grothe, K.L. Dickson, and D.K. Reed-Judkins (eds.), pp. 245-270. SETAC Press, Pensacola, FL.

Barbour, M.T., J. Gerritsen, G.E. Griffith, R. Frydenborg, E. McCarron, J.S. White, and M.L. Bastian. 1996b. A framework for biological criteria for Florida streams using benthic macroinvertebrates. *Journal of the North American Benthological Society* 15:185-211.

Barbour, M.T. 1997. The re-invention of biological assessment in the U.S. *Human and Ecological Risk Assessment* 3(6):933-940.

Barbour, M.T., J. Gerritsen, B.D. Snyder, and J.B. Stribling. 1999. *Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish. Second Edition.* EPA 841-B-99-002. U.S. EPA, Office of Water, Washington, DC.

Barbour, M.T. and C.O. Yoder. Technical Guidelines: Technical Elements of a Bioassessment Program. Unpublished manuscript.

Bendellyoung, L.I., K.E. Bennett, A. Crowe, C.J. Kennedy, A.R. Kermode, M.M. Moore, A.L. Plant, and A. Wood. 2000. Ecological characteristics of wetlands receiving an industrial effluent. *Ecological Applications* 10:310-322.

Benke, A.C. and C.E. Cushing (eds.). 2005. *Rivers of North America*. Academic Press, New York.

Booth, D.B., J.R. Karr, S. Schauman, K.P. Kinrad, S.A. Morley, M.G. Larson, and S.J. Burges. 2004. Reviving urban streams: land use, hydrology, biology, and human behavior. *Journal of the American Water Resources Association* 40:1351-1364.

Brinely, F.J. 1942. Biological studies, Ohio River pollution survey. I. Biological zones in a polluted stream. *Sewage Works Journal* 14(1):147-152.

Brinkhurst, R. 1993. Future directions in freshwater biomonitoring. In *Freshwater Biomonitoring and Benthic Macroinvertebrates*, D.H. Rosenberg and V. H. Resh (eds.), pp. 442-460. Chapman and Hall, New York.

Brinson, M.M. and A.I. Malvarez. 2002. Temperate freshwater wetlands: types, status, and threats. *Environmental Conservation* 29:115-133.

Brönmark, C. and L-A. Hansson. 2002. Environmental issues in lakes and ponds: current state and perspectives. *Environmental Conservation* 29:290-306.

Bryce, S.A., D.P. Larsen, R.M. Hughes, and P.R. Kaufmann. 1999. Assessing relative risks to aquatic ecosystems: a mid-Appalachian case study. *Journal of the American Water Resources Association* 35:23-36.

Bryce, S.A., R.M. Hughes, and P.R. Kaufman. 2002. Development of a bird integrity index: using bird assemblages as indicators of riparian condition. *Environmental Management* 30:294-310.

Bunn S.E. and Arthington A.A. 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. *Environmental Management* 30:492-507.

Cairns, J. Jr. 1977. Quantification of biological integrity. In *The Integrity of Water*, R.K. Ballentine and L.J. Guarraia (eds.), pp. 171-187. Proceedings of a Symposium, March 10-12, 1975. U.S. Environmental Protection Agency, Washington, DC.

Cairns, J. Jr. 1981. Biological monitoring part VI-future needs. *Water Research* 15:941-952.

Cairns, J. Jr. and J. R. Pratt. 1993. A history of biological monitoring using benthic macroinvertebrates. In *Freshwater Biomonitoring and Benthic Macroinvertebrates*, D.M. Rosenberg and V.H. Resh (eds.), pp. 10-27. Chapman & Hall, New York.

Cairns, J. Jr., P. V. McCormick, and R. R. Niederlehner. 1993. A proposed framework for developing indicators of ecosystem health. *Hydrobiologia* 263:1-44.

Contreras-Balderas, S., R. J. Edwards, M. L. Lozano-Vilano and M. E. García-Ramírez. 2002. Fish biodiversity changes in the Lower Rio Grande/Rio Bravo, 1953-1996. *Reviews in Fish Biology and Fisheries* 12(2):219-240.

Campbell, M.S.A. 1939. Biological indicators of intensity of stream pollution. *Sewage Works Journal* 11(1):123-127.

Canada and the United States. 2001. State of the Great Lakes 2001. Toronto and Chicago. 82 p.

Carlson, R.E. 1992. Expanding the trophic state concept to identify non-nutrient limited lakes and reservoirs. In *Proceedings, National Conference on Enhancing the States' Lake Management Programs*, Chicago, IL, 1991, pp. 59-71. North American Lake Management Society.

Carpenter, S., N.F. Caraco, D.L. Correll, R.W. Howarth, A.N. Sharpley, and V.H. Smith. 1998. Nonpoint pollution of surface waters with phosphorous and nitrogen. *Issues in Ecology*, No. 3.

Cech Jr., J.J., B.W. Wilson, and D.G. Crosby. 1998. *Multiple stressors in ecosystems*. Lewis Publishers, 202p.

Chaloud, D. J. and M. S. Nash. 2001. Using cononical correlation to detect association of landscape metrics with water biological and chemical properties in Savannah River Basin. Presented at Above & Beyond 2001, An EPA Remote Sensing Conference, Las Vegas, NV, March 20-21, 2001.

Courtemanch, D.L. 1984. A closing artificial substrate device for sampling benthic macroinvertebrates in deep rivers. *Freshwater Invertebrate Biology* 3(3):143-146.

Courtemanch, D.L., S.P. Davies, and E.B. Laverty. 1989. Incorporation of biological information in water quality planning. *Environmental Management* 13(1):35-41.

Courtemanch, D.L. 1995. Merging the science of biological monitoring with water resource management policy: Criteria development. In *Biological assessment and criteria: Tools for water resource planning and decision making*, W.S. Davis and T.P. Simon (eds.), pp. 315-325. Lewis Publishers, Boca Raton, FL.

Courtney, L. A. and W. H. Clements. 2000. Sensitivity to acidic pH in benthic invertebrate assemblages with different histories of exposure to metals. *Journal of the North American Benthological Society* 19:112-127.

Croonquist, M. and R. P. Brooks. 1993. Effects of habitat disturbance on bird communities in riparian corridors. *Journal of Soil and Water Conservation* 48:65-70.

Danz, N.P., R.R. Regal, G.J. Niemi, V.J. Brady, T. Hollenhorst, L.B. Johnson, G.E. Host, J.M. Hanowski, C.A. Johnston, T. Brown, J. Kingston, and J.R. Kelly. 2005. Environmentally stratified sampling design for the development of Great Lakes environmental indicators. *Environmental Monitoring and Assessment*.

Davies, P.E. 2000. Development of a national river bioassessment system (AusRivAS) in Australia. In *Assessing the Biological Quality of Fresh Waters*, J.F. Wright, D.W. Sutcliffe and M.T. Furse (eds.), pp. 113-124. Freshwater Biological Association, Ambleside.

Davies, S.P. 1987. Benthic macroinvertebrate response to pollution abatement in the Penobscot River, Maine. MS thesis, University of Maine, Orono, ME. 83 pp.

Davies, S.P., L. Tsomides, D.L. Courtemanch and F. Drummond. 1995. *Maine biological monitoring and biocriteria development program*. DEP-LW108, Maine Department of Environmental Protection, Augusta, ME. P.61.

Davies, S.P., L. Tsomides, J.L. DiFranco, and D.L. Courtemanch. 1999. *Biomonitoring Retrospective: Fifteen Year Summary for Maine Rivers and Streams*. MDEP (DEP LW1999-26).

Davies, S.P. and L. Tsomides. 2002. *Methods for Biological Sampling and Analysis of Maine's Rivers and Streams*. MDEP (DEP LW0387-B2002).

Davies, S.P., F. Drummond, D.L. Courtemanch, L. Tsomides. Probabilistic Models Based on Expert Judgment Protocols to Assess Attainment of Tiered Aquatic Life Uses in Maine Rivers and Streams. Unpublished manuscript.

Davies, S.P. and S.K. Jackson. In press. The Biological Condition Gradient: A descriptive model for interpreting change in aquatic ecosystems. *Ecological Applications*.

Davis, W.S. 1995. Biological assessment and criteria: Building on the past. In *Biological assessment and criteria: Tools for water resource planning and decision making*, W.S. Davis and T.P. Simon (eds.), pp. 15-29. Lewis Publishers, Boca Raton, FL.

Day, J.W., C.A.S. Hall, W.M. Kemp, and A. Yanez-Arenciba. 1989. *Estuarine Ecology*. John Wiley & Sons, New York.

DeShon, J.D. 1995. Development and application of the invertebrate community index (ICI). In *Biological assessment and criteria: Tools for water resource planning and decision making*, W.S. Davis and T.P. Simon (eds.), pp. 217-243. Lewis Publishers, Boca Raton, FL.

Diamond, J.M. and V.B. Serveiss. 2001. Identifying sources of stress to native aquatic fauna using a watershed ecological risk assessment framework. *Environmental Science and Technology* 35:4711-4718.

Dixit, S. S., J. P. Smol, D. F. Charles, R. M. Hughes, S. G. Paulsen, and G. B. Collins. 1999. Assessing water quality changes in the lakes of the northeastern United States using sediment diatoms. *Canadian Journal of Fisheries and Aquatic Sciences* 56:131-152.

Doudoroff, P. and C.E. Warren. 1957. Biological indices of water pollution with special reference to fish populations. Biological Problems in Water Pollution, U.S. Public Health Service, Robert A. Taft Sanitary Engineering Center, Cincinnati, OH. 144-163.

Dynesius, M. and C. Nilsson. 1994. Fragmentation and flow regulation of river systems in the northern third of the world. *Science* 266:753-762.

Ellis, M.M. 1937. Detection and measurement of stream pollution. *Bulletin of the Bureau of Fisheries* 48:365-437.

Emery, E. B., T. P. Simon, F. H. McCormick, P. A. Angermeier, J. E. DeShon, C. O. Yoder, R. E. Sanders, W. D. Pearson, G. D. Hickman, R. J. Reash, and J. A. Thomas. 2003. Development of a Multimetric Index for Assessing the Biological Condition of the Ohio River. *Transactions of the American Fisheries Society* 132:791-808.

Fausch, K.D., J.R. Karr, and P.R. Yant. 1984. Regional application of an index of biotic integrity based on stream fish communities. *Transactions of the American Fisheries Society* 113:39-55.

Fausch, K.D., J. Lyons, P. L. Angermeier, and J. R. Karr. 1990. Fish communities as indicators of environmental degradation. *American Fisheries Society Symposium* 8:123-144.

Fausch, K.D., C.E. Torgersen, C.V. Baxter, and H.W. Li. 2002. Landscapes to riverscapes: bridging the gap between research and conservation of stream fishes. *BioScience* 52:483-498.

Federal Water Pollution Control Act, as amended by the Clean Water Act of 1977: [Commonly referred to as Clean Water Act] [Enacted by Public Law 92-500, October 18, 1972, 86 Stat. 816; 33 U.S.C. 1251 et seq.; Amended by PL 93-207, December 28, 1973, and PL 93-243, January 2, 1974; PL 93-592, January 2, 1975; PL 94-238, March 23, 1976; PL 94-273, April 21, 1976; PL 94-558, October 19, 1976; PL 95-217, December 28, 1977; PL 95-576, November 2, 1978; PL 96-148, December 16, 1979; PL 96-478, PL 96-483, October 21, 1980; PL 96-510, December 11, 1980; PL 96-561, December 22, 1980; PL 97-35, August 13, 1981; PL 97-117, December 29, 1981; PL 97-164, April 2, 1982; PL 97-440, January 8, 1983; Amended by PL 100-4, February 4, 1987].

Fitzgerald, D. G., R. P. Lanno, and D. G. Dixon. 1999. A comparison of a sentinel species evaluation using creek chub (*Semotilus atromaculatus mitchill*) to a fish community evaluation for the initial identification of environmental stressors in small streams. *Ecotoxicology* 8:33-48.

Fitzhugh, T.W. and B.D. Richter. 2004. Quenching urban thirst: growing cities and their impacts on freshwater ecosystems. *BioScience* 54: 741-754.

Fitzpatrick, F. A., B. C. Scudder, B. N. Lenz, and D. J. Sullivan. 2001. Effects of multi-scale environmental characteristics on agricultural stream biota in eastern Wisconsin. *Journal of the American Water Resources Association* 37:1489-1507.

Freeman, M.C., E.R. Irwin, N.M. Burkhead, B.J. Freeman, and H. L. Bart, Jr. 2005. Status and Conservation of the Fish Fauna of the Alabama River System. In *Historical Changes in Large River Fish Assemblages of the Americas*, J.N. Rinne, R. M. Hughes, and B. Calamusso (eds.). American Fisheries Society Symposium 45, Bethesda, Maryland.

Frey, D.G. 1977. Biological Integrity, a Historical Approach. In *The Integrity of Water*, R.K. Ballentine and L.J. Guarraia (eds.), pp. 127-140. Proceedings of a Symposium, March 10-12, l975. U.S. EPA, Office of Water and Hazardous Materials, Washington, DC.

Frissell, C.A. 1993. Topology of extinction and endangerment of native fishes in the Pacific Northwest and California (USA). *Conservation Biology* 7:342-354.

Furse, M.T., D. Moss, J.F. Wright, and P.D. Armitage. 1984. The influence of seasonal and taxonomic factors on the ordination and classification of running-water sites in Great Britain and on the prediction of their macro-invertebrate communities. *Freshwater Biology* 14:257-80.

Gakstatter, J., J.R. Gammon, R.M. Hughes, L. Ischinger, M. Johnson, J.R. Karr, T. Murphy, T.M. Murray, and T. Stewart. 1981. *A recommended approach for determining biological integrity in flowing waters*. U.S. EPA, Corvallis, OR. 26 pp.

Gammon, J.R. 1976. *The fish population of the middle 340km of the Wabash River*. Purdue University Water Resources Research Center, LaFayette, IN. Technical Report 86.

Gammon, J.R., A. Spacie, J.L Hamelink, and R.L. Kaesler. 1981. Role of electrofishing in assessing environmental quality of the Wabash River. In *Ecological Assessments of Effluent Impacts on Communities of Indigenous Aquatic Organisms*, J. Bates and C.I. Weber (eds.), pp. 307-324. ASTM STP 730. Philadelphia, PA.

Ganasan, V. and R. M. Hughes. 1998. Application of an index of biological integrity (IBI) to fish assemblages of the rivers Khan and Kshipra (Madhya Pradesh), India. *Freshwater Biology* 40:367-383.

Gaufin, A.R. and C.M. Tarzwell. 1953. Discussion of R. Patrick's paper, "Aquatic organisms as an aid in solving waste disposal problems." *Sewage and Industrial Wastes* 25(2):214-217.

Gerritsen, J., M.T. Barbour, and K. King. 2000. Apples, oranges and ecoregions: On determining pattern in aquatic assemblages. *Journal of North American Benthological Society* 19:487-496.

Gerritsen, J. and E.W. Leppo. 2004. *Tiered aquatic life use development for New Jersey*. Prepared by Tetra Tech, Inc. for U.S. EPA Office of Water, EPA Region 2, and New Jersey DEP.

Gessner, M.O. and E. Chauvet. 2002. A case for using litter breakdown to assess functional stream integrity. *Ecological Applications* 12: 498-510.

Gibson, G.R., M.T. Barbour, J.B. Stribling, J. Gerritsen, and J.R. Karr. 1996. *Biological criteria: Technical guidance for streams and small rivers (revised edition)*. EPA/822/B/96/001. U.S. EPA, Office of Water, Washington, DC.

Graf, W. L. 2001. Damage control: restoring the physical integrity of America's rivers. *Annals of the Association of American Geographers* 91:1-27.

Greenberg L, P. Svendsen, and A. Harby. 1996. Availability of microhabitats and their use by brown trout (*Salmo trutta*) and grayling (*Thymallus thymallus*) in the River Vojman, Sweden. *Regulated Rivers: Research & Management* 12: 287–303.

Griffith, J.A., E.A. Martinko, J.L. Whistler, and K.P. Price. 2002. Interrelationships among landscapes, NDVI, and stream water quality in the U.S. Central Plains. *Ecological Applications* 12(6): 1702-1718.

Grove, R.H. 1995. *Green imperialism: colonial expansion, tropical island edens, and the origins of environmentalism, 1600-1860*. Cambridge University Press, Cambridge, United Kingdom.

Halliwell, D. B., R. W. Langdon, R. A. Daniels, J. P. Kurtenbach, and R. A. Jacobson. 1998. Classification of freshwater fish species of the Northeastern United States for use in the development of indices of biological integrity, with regional applications. In *Assessing the sustainability and biological integrity of water resources using fish communities*, T. P. Simon (ed.), pp. 301-337. CRC Press, Boca Raton, FL.

Harding, J.S., E.F. Benfield, P.V. Bolstad, G.S. Helfman, and E.B.D. Jones III. 1998. Stream biodiversity: the ghost of land use past. *Proceedings of the National Academy of Science* USA 95: 14843-14847.

Hawkins, C.P. and M.R. Vinson. 2000. Weak correspondence between landscape classifications and stream invertebrate assemblages: implications for bioassessment. *Journal of North American Benthological Society* 19: 501-517.

Hawkins, C.P., R. Norris, J. Gerritsen, R.M. Hughes, S.K. Jackson, R.K. Johnson, and R.J. Stevenson. 2000. Evaluation of the use of landscape classifications for the prediction of freshwater biota: synthesis and recommendation *Journal of North American Benthological Society* 19:541-556.

Helmsley-Flint, B. 2000. Classification of the biological quality of rivers in England and Wales. In *Assessing the Biological Quality of Fresh Waters*, J.F. Wright, D.W. Sutcliffe and M.T. Furse (eds.), pp. 55-70. Freshwater Biological Association, Ambleside, UK.

Herlihy, A.T., R.M. Hughes, and J.C. Sifneos. In Press. National clusters of fish species assemblages in the conterminous United States and their relationship to existing landscape classification schemes. In. *Influences of landscapes on stream habitats and biological assemblages*, R.M. Hughes, L. Wang, and P.W. Seelbach (eds.), Pages xx-xx. American Fisheries Society Symposium, Bethesda, Maryland.

Herricks, E.E. and D.J. Schaeffer. 1985. Can we optimize biomonitoring? *Environmental Management* 9:487-492.

Hilsenhoff, W.L. 1987. An improved biotic index of organic stream pollution. *Great Lakes Entomologist* 20: 31-39.

Holden, P. B. and C. B. Stalnaker. 1975. Distribution and abundance of mainstem fishes of the middle and upper Colorado River Basins, 1967-1973. *Transactions of the American Fisheries Society* 104:217-231.

Hughes, R.M. 1985. Use of watershed characteristics to select control streams for estimating effects of metal mining wastes on extensively disturbed streams. *Environmental Management* 9:253-262.

Hughes, R.M., D.P. Larsen, and J.M. Omernik. 1986. Regional reference sites: a method for assessing stream potential. *Environmental Management* 10:629-635.

Hughes, R. M. and J. R. Gammon. 1987. Longitudinal changes in fish assemblages and water quality in the Willamette River, Oregon. *Transactions of the American Fisheries Society* 116:196-209.

Hughes, R.M. and D.P. Larsen. 1988. Ecoregions: an approach to surface water protection. *Journal of the Water Pollution Control Federation* 60: 486-493.

Hughes, R. M. 1994. Defining acceptable biological status by comparing with reference conditions. In *Biological assessment and criteria: tools for water resource planning and decision making*, W. S. Davis and T. P. Simon (eds.), pp. 31-47. CRC Press, Boca Raton, FL.

Hughes, R. M., P. R. Kaufmann, A. T. Herlihy, T. M. Kincaid, L. Reynolds, and D. P. Larsen. 1998. A process for developing and evaluating indices of fish assemblage integrity. *Canadian Journal of Fisheries and Aquatic Sciences* 55:1618-1631.

Hughes, R. M., and T. Oberdorff. 1998. Applications of IBI concepts and metrics to waters outside the United States and Canada. In *Assessing the sustainability and biological integrity of water resources using fish communities*, T. P. Simon (ed.), pp. 79-93. CRC Press, Boca Raton, FL.

Hughes, R.M., S. Howlin, and P.R. Kaufmann. 2004. A biointegrity index (IBI) for coldwater streams of western Oregon and Washington. *Transactions of the American Fisheries Society* 133:1497-1515.

Hughes, R. M., S. A. Bryce, and D. Drake. Use of a Generalized Stressor Gradient for Comparing Reference Conditions of USA Surface Waters. Unpublished manuscript.

Intergovernmental Task Force on Monitoring Water Quality (ITFM). 1992. *Ambient Water-Quality Monitoring in the United States. First Year Review, Evaluation, and Recommendations*. Interagency Advisory Committee on Water Data, U.S. Geological Survey, Reston, Virginia.

Intergovernmental Task Force on Monitoring Water Quality (ITFM). 1995. *The strategy for improving water-quality monitoring in the United States: Final report of the Intergovernmental Task Force on Monitoring Water Quality*. U.S. Geological Survey, Reston, VA.

Jackson, J.B.C., M.X. Kirby, W.H. Berger, K.A. Bjorndahl, L.W. Botsford, B.J. Bourque, R.H. Bradbury, R. Cooke, J. Erlandson, J.A. Estes, T.P. Hughes, S. Kidwell, C.B. Lange, H.S. Lenihan, J.M. Pandolfi, C.H. Peterson, R.S. Steneck, M.J. Tegner, and R.R. Werner. 2001. Historical overfishing and the recent collapse of coastal ecosystems. *Science* 293: 629-638.

Jenks, G.F. and F.C. Caspall. 1971. Error on choroplethic maps - definition, measurement, reduction. *Annals of the Association of American Geographers* 61(2):217-244.

Johnson, R.K., T. Wiederholm, and D.M. Rosenberg. 1993. Freshwater biomonitoring using individual organisms, populations, and species assemblages of benthic macroinvertebrates. In *Freshwater Biomonitoring and Benthic Macroinvertebrates*, D.M. Rosenberg and V.H. Resh (eds.), pp. 40-125. Chapman & Hall, New York.

Johnson, W.C. 1994. Woodland Expansion in the Platte River, Nebraska: Patterns and Causes. *Ecological Monographs* 64:45-84.

Jongman, R.H.G., C.J.F. ter Braak and O.F.R. Tongeren (eds.). 1987. *Data Analysis in Community and Landscape Ecology*. Pudoc, Wageningen, NL.

Karr, J.R. 1981. Assessment of biotic integrity using fish communities. *Fisheries* 6(6):21 27.

Karr, J.R. and D.R. Dudley. 1981. Ecological perspective on water quality goals. *Environmental Management* 5:55-68.

Karr, J.R., 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* 5.

Karr, J.R. 1987. Biological monitoring and environmental assessment: A conceptual framework. *Environmental Management* 11:249-256.

Karr, J.R., P.R. Yant, K.D. Fausch, and I.J. Schlosser. 1987. Spatial and temporal variability of the Index of Biotic Integrity in three midwestern streams. *Transactions of the American Fisheries Society* 116:1-11.

Karr, J.R. 1990. Biological integrity and the goal of environmental legislation: lessons from conservation biology. *Conservation Biology* 4:244-250.

Karr, J.R. 1998. Rivers as sentinels: Using the biology of rivers to guide landscape management. In. *River Ecology and Management: Lessons from the Pacific Coastal Ecosystems*, R. J. Naiman and R. E. Bilby (eds.), pp. 502-528. Springer, NY.

Karr, J.R. and E. W. Chu. 1999. *Restoring Life In Running Waters: Better Biological Monitoring*. Island Press, Washington, DC.

Karr, J.R. 2000. Health, integrity, and biological assessment: The importance of whole things. In *Ecological Integrity: Integrating Environment, Conservation, and Health*, D. Pimentel, L. Westra, and R. F. Noss (eds.), pp. 209-226. Island Press, Washington, DC.

Karr, J.R. and E.W. Chu. 2000. Sustaining living rivers. *Hydrobiologia* 422:1-14.

Karr, J.R. and E. M. Rossano. 2001. Applying public health lessons to protect river health. *Ecology and Civil Engineering* 4: 3-18.

Karr, J.R. and C. O. Yoder. 2004. Biological assessment and criteria improve total maximum daily load decision making. *Journal of Environmental Engineering* 130: 594-604.

Kaufmann, P.R., and R.M. Hughes. In Press. Geomorphic and anthropogenic influences on fish and amphibians in Pacific Northwest coastal streams. In *Influences of landscapes on stream habitats and biological assemblages*, R. M. Hughes, L. Wang, and P. Seelbach (eds.), pages xx-xx. American Fisheries Society Symposium xx. Bethesda, Maryland.

Kerans, B. L., and J. R. Karr. 1994. A benthic index of biotic integrity (B-IBI) for rivers of the Tennessee Valley. *Ecological Applications* 4:768-785.

Klemm, D.J., P.A. Lewis, F. Fulk, and J.M. Lazorchak. 1990. *Macroinvertebrate field and laboratory methods for evaluating the biological integrity of surface water*. EPA/a600/4-90/030. U.S. Environmental Protection Agency, Cincinntti, OH, 256 pp.

Klauda, R., P. Kazyak, S. Stranko, M. Southerland, N. Roth, and J. Chaillou. 1998. Maryland Biological Stream Survey: A state agency program to assess the impact of anthropogenic stresses on stream habitat quality and biota. *Environmental Monitoring and Assessment* 51: 299-316.

Krumholz, L.A. and W.L. Minckley. 1964. Changes in the fish population in the upper Ohio River following temporary pollution abatement. *Transactions of the American Fisheries Society* 93(1)1-5.

Kuzelka, R.D., C.A. Flowerday, R.N. Manley, B.C. Rundquist, S.J. Herrin, and C.A. Flowerday (eds.). 1993. *Flat water: A history of Nebraska and its water*. Resource Report No. 12, Conservation and Survey Division, Institute of Agriculture and Natural Resources, University of Nebraska – Lincoln.

Ladson, A. R., L. White, J. A. Doolan, B. L. Finlayson, B. T. Hart, S. Lake, and J. W. Tilleard. 1999. Development and testing of an Index of Stream Condition for waterway management in Australia. *Freshwater Biology* 41(2):453-468.

Lammert, M., and J. D. Allan. 1999. Assessing biotic integrity of streams: effects of scale in measuring the influence of land use/cover and habitat structure on fish and macroinvertebrates. *Environmental Management* 23:257-270.

Lattin, P. D., P. J. Wigington Jr., T. J. Moser, B. E. Peniston, D. R. Lindeman, and D. R. Oetter. 2004. Influence of remote sensing imagery source on quantification of riparian land cover-land use. *Journal of the American Water Resources Association* 40:215-227.

Leeson, M. A. 1885. *History of Montana 1739-1885*. Warner, Beers & Co., Chicago.

Legendre, P. and L. Legendre. 1998. *Numerical Ecology*. Second English Edition. Elsevier, Amsterdam.

Leonard, P. M., and D. J. Orth. 1986. Application and testing of an index of biotic integrity in small coolwater streams. *Transactions of the American Fisheries Society* 115:401-414.

Lorentz, C. M., G. M. Van Dijk, A. G. M. Van Hattum, and W. P. Cofino. 1997. Concepts in river ecology: implications for indicator development. *Regulated Rivers: Research and Management* 13:501-516.

Lubinski, K. and C. Theiling. 1999. Assessments and forecasts of the ecological health of the Upper Mississippi River System Floodplain Reaches. Ch. 2. in *Ecological Status and Trends of the Upper Mississippi River System 1998*, Lubinski and Theiling, eds. United States Geological Survey, Report of the Long Term Resource Monitoring Program, LTRMP 99-T001, LaCrosse, WI.

Ludwig, J.A. and J.F. Reynolds. 1988. *Statistical Ecology*. John Wiley & Sons, New York.

Malmqvist, B. and S.D. Rundle. 2002. Threats to the running water ecosystems of the world. *Environmental Conservation* 29: 134-153.

Margalef, R. 1963. On certain unifying principles in ecology. *American Naturalist* 97:357-374.

Margalef, R. 1981. Stress in ecosystems: a future approach. In *Stress effects on natural ecosystems*, G.W. Barrett and R. Rosenberg (eds.), pp. 281-289. Wiley, London, UK.

Martinez, M. E. 1998. What is problem-solving? *Phi Delta Kappan* 79:605-610.

McCormick, F.H., D.V. Peck, and D.P. Larsen. 2000. A comparison of ecological classification hierarchies for Mid-Atlantic stream fish assemblages. *Journal of North American Benthological Society* 19:385-404.

McCormick, F.H., R. M. Hughes, P. R. Kaufmann, D.V. Peck, J. L. Stoddard, and A.T. Herlihy. 2001. Development of an index of biotic integrity for the Mid-Atlantic Highlands. *Transactions of the American Fisheries Society* 130: 857-877.

Mebane, C. A., T. R. Maret, and R. M. Hughes. 2003. An index of biological integrity (IBI) for Pacific Northwest rivers. *Transactions of the American Fisheries Society* 132:239-261.

Meyer W.B., Turner B.L., II (eds.). 1994. *Changes in Land-Use and Land-Cover: A global perspective*. Cambridge University Press, Cambridge, UK.

Miller, R.R., J.D. Williams, and J.E. Williams. 1989. Extinctions of North American fishes during the past century. *Fisheries* 14:22-38.

Mills, H.B., W.C. Starrett, and F.C. Bellrose. 1966. Man's effect on the fish and wildlife of the Illinois River. *Illinois Natural History Survey Biological Notes* 57.

Miltner, R.J., D. White, and C.O. Yoder. 2003. The biotic integrity of streams in urban and suburbanizing landscapes. *Landscape and Urban Planning* 69(2004): 87-100.

Morley, S. A. and J. R. Karr. 2002. Assessing and restoring the health of urban streams in the Puget Sound Basin. *Conservation Biology* 16:1498-1509.

Moss, D., M.T. Furse, J.F. Wright, and P.D. Armitage. 1987. The prediction of the macroinvertebrate fauna of unpolluted running-water sites in Great Britain using environmental data. *Freshwater Biology* 17:41-52.

Moyle, P. B. 1986. Fish introductions into North America: patterns and ecological impact. In *Ecology of biological invasions of North America and Hawaii*, H. A. Mooney and J. A. Drake (eds.), pp. 27-43. Springer, New York.

Myers, R.A. and B. Worm. 2003. Rapid worldwide depletion of predatory fish communities. *Nature* 423:280-283.

Naiman and Decamps (eds.). 1990. *The ecology and management of aquatic-terrestrial ecotones*. Parthenon Publishing, Carnforth, UK.

Nash, M. S. and D.J. Chaloud. 2002. *Multivariate Analyses (Canonical Correlation Analysis and Partial Least Square, PLS) to Model and Assess the Association of Landscape Metrics to Surface Water Chemical and Biological Properties using Savannah River Basin Data*. EPA/600/R-02/091. United States Environmental Protection Agency, Washington, DC.

National Research Council. 1992. *Restoration of Aquatic Ecosystems*. National Academy Press, Washington DC.

National Research Council. 2001. *Assessing the TMDL Approach to Water Quality Management*. Water Science and Technology Board, Division on Earth and Life Studies. National Academy Press, Washington, DC.

Nehlsen, W., J. E. Williams, and J. A. Lichatowich. 1991. Pacific salmon at the crossroads: stocks at risk from California, Oregon, Idaho, and Washington. *Fisheries* 16(2):4-21.

Nerbonne B.A. and B. Vondracek. 2001. Effects of local land use on physical habitat, benthic macroinvertebrates, and fish in the Whitewater River, Minnesota, USA. *Environmental Management* 28:87-99.

Nilsson C., A. Ekblad, M. Gardfjell, and B. Carlberg. 1991. Long-term effects of river regulation on river margin vegetation. *Journal of Applied Ecology* 28:968-987.

Odum, E.P., J.T. Finn, and E.H. Franz. 1979. Perturbation theory and the subsidy-stress gradient. *BioScience* 29:349-352.

Odum, E.P. 1985. Trends expected in stressed ecosystems. *BioScience* 35:419-422.

Ohio Administrative Code 3745-1-07 c. 1978.

Ohio Department of Natural Resources (Ohio DNR). 1960. *Gazetteer of Ohio streams*. Ohio Water Plan Inventory Report Number 12. Division of Water, Columbus, Ohio. 179 pp.

Ohio Environmental Protection Agency (Ohio EPA). 1980. *Manual of surveillance methods and quality assurance practices*. Office of Wastewater Pollution Control, Columbus, Ohio.

Ohio Environmental Protection Agency (Ohio EPA). 1981. *5-year surface water monitoring strategy, 1982-1986*. Office of Wastewater Pollution Control, Division of Surveillance and Standards, Columbus, Ohio. 52 pp. + appendices.

Ohio Environmental Protection Agency (Ohio EPA). 1987. *Biological criteria for the protection of aquatic life: volumes I-III*. Ohio Environmental Protection Agency, Columbus, Ohio.

Ohio Environmental Protection Agency (Ohio EPA). 1988. *Ohio water quality inventory – 1988 305(b) report, volume I and executive summary*. E.T. Rankin, C.O. Yoder and D.A. Mishne (eds.). Ohio EPA, Division of Water Quality Monitoring and Assessment, Columbus, OH.

Ohio Environmental Protection Agency (Ohio EPA). 1989a. *Biological criteria for the protection of aquatic life: Volume III. Standardized biological field sampling and laboratory methods for assessing fish and macroinvertebrate communities.* Division of Water Quality Monitoring and Assessment, Columbus, Ohio.

Ohio Environmental Protection Agency (Ohio EPA). 1989b. *Addendum to biological criteria for the protection of aquatic life: Volume II. Users manual for biological field assessment of Ohio surface waters*. Division of Water Quality Monitoring and Assessment, Surface Water Section, Columbus, Ohio.

Ohio Environmental Protection Agency (Ohio EPA). 1990. *The use of biocriteria in the Ohio EPA surface water monitoring and assessment program.* Ohio Environmental Protection Agency, Division of Water Quality Planning and Assessment, Columbus, Ohio.

Ohio Environmental Protection Agency (Ohio EPA). 1992. *Ohio water resource inventory. Volume I: Summary, status, and trends*. E. Rankin, C. Yoder, and D. Mishne (eds.). Ohio Environmental Protection Agency, Division of Water Quality Monitoring and Assessment, Columbus, Ohio.

Ohio Environmental Protection Agency (Ohio EPA). 1996. *Justification and Rationale for Revisions to the Dissolved Oxygen Criteria in the Ohio Water Quality Standards*, OEPA Technical Bulletin MAS/1995-12-5, State of Ohio Environmental Protection Agency, Division of Surface Water, Columbus, Ohio.

Ohio Environmental Protection Agency (Ohio EPA). 1998. *Ohio Water Resource Inventory, Volume I: Summary, Status and Trends*. E. T. Rankin, C. O. Yoder, and D.Mishne, (eds.). Division of Surface Water, Ecological Assessment Section. Columbus, Ohio.

Ohio Environmental Protection Agency (Ohio EPA). 1999a. *Associations between nutrients, habitat, and the aquatic biota of Ohio's rivers and streams*. Technical Bulletin MAS/1999-1-1, Division of Surface Water, Monitoring and Assessment Section, Columbus, Ohio.

Ohio Environmental Protection Agency (Ohio EPA). 1999b. *Total maximum daily load TMDL team report*. Division of Surface Water, Columbus, Ohio.

Ohio Environmental Protection Agency (Ohio EPA). 2000. *Ohio Water Resource Inventory, Volume I: Summary, Status and Trends*. E. T. Rankin, C. O. Yoder, and D.Mishne, (eds.). Division of Surface Water, Ecological Assessment Section. Columbus, Ohio.

Ohio Environmental Protection Agency (Ohio EPA). 2003. *Ohio EPA's primary headwater habitat project: key findings*. Fact sheet. Division of Surface Water, Columbus, Ohio. 2 pp.

Ohio Environmental Protection Agency (Ohio EPA). 2004. *Total maximum daily loads for the Stillwater River Basin*. Division of Surface Water, Columbus, Ohio.

Omernik, J.M. 1987. Ecoregions of the conterminous United States. *Annals of the Association of American Geographers* 77:118-125.

Omernik, J.M. 1995. Ecoregions: A spatial framework for environmental management. In *Biological assessment and criteria: Tools for water resource planning and decision making*, W.S. Davis and T.P. Simon (eds.), pp. 49-62. Lewis Publishers, Boca Raton, FL.

Omernik, J.M. 2003. The misuse of hydrologic unit maps for extrapolation, reporting and ecosystem management. *Journal of the American Water Resources Association* 39(3):563-573.

Organization for Economic Cooperation and Development (OECD). 1993. Core set of indicators for environmental performance reviews. *Environmental Monographs* 83.

Patrick, R.M. 1950. Biological measure of stream conditions. *Sewage and Industrial Wastes* 22:926-938.

Patrick, R.M. 1953. Aquatic organisms as an aid in solving waste disposal problems. *Sewage and Industrial Wastes* 25(2):210-214.

Paul, M.J. and J.L. Meyer. 2001. The ecology of urban streams. *Annual Review of Ecology and Systematics* 32:333-365

Pianka, E. R. 1970. On r- and k-selection. *American Naturalist* 104:592-597.

Plafkin, J.L., M.T. Barbour, K.D. Porter, S.K. Gross, and R.M. Hughes. 1989. *Rapid bioassessment protocols for use in streams and rivers: benthic macroinvertebrates and fish*. EPA 440-4-89-001. Office of Water Regulations and Standards, U.S. Environmental Protection Agency, Washington, DC.

Poff, N.L. and J.D. Allan. 1995. Functional organization in stream fish assemblages in relation to hydrologic variability. *Ecology* 76:606-627.

Poff, N.L., J.D. Allan, M.B. Bain, J.R. Karr, K.L. Prestegaard, B.D. Richter, R.E. Sparks, and J.C. Stromberg. 1997. The natural flow regime: a paradigm for river conservation and restoration. *BioScience* 47:769-784.

Poff, N.L. and A.D. Huryn. 1998. Multi-scale determinants of secondary production in Atlantic salmon (*Salmo salar*) streams. *Canadian Journal of Fisheries and Aquatic Sciences* 55(Suppl. 1):201-217.

Potter, K.M., F.W. Cubbage, and R.H. Schaberg. 2005. Multiple-scale landscape predictors of benthic macroinvertebrate community structure in North Carolina. *Landscape and Urban Planning* 71:77-90.

Pretty, J.L., S.S.C. Harrison, D.J. Shepherd, C. Smith, A.G. Hildrew, and R.D. Hey. 2003. River rehabilitation and fish populations: assessing the benefit of instream structures. *Journal of Applied Ecology* 40:251-265.

Pulliam, H. R. 1988. Sources, sinks, and population regulation. *American Naturalist* 132: 652-661.

Quinn, J.M. 2000. Effects of pastoral development. In *New Zealand Stream Invertebrates: Ecology and Implications for Management*, K.J. Collier and M.J. Winterbourn (eds.), pp. 208-229. Caxton, Christchurch, NZ.

Rabeni, C.F. 1977. Benthic macroinvertebrate communities of the Penobscot River, Maine, with special reference to their role as water quality indicators. Ph.D thesis, University of Maine, Orono. 169 pp.

Rabeni, C. F., S. P. Davies, and K. E. Gibbs. 1988. Benthic macroinvertebrate response to pollution abatement: structural changes and functional implications. *Water Resources Bulletin* 21:489-497

Rankin, E.T. 1989. *The qualitative habitat evaluation index (QHEI), rationale, methods, and application*. Ohio Environmental Protection Agency, Division of Water Quality Planning and Assessment, Ecological Assessment Section, Columbus, OH.

Rankin, E.T. 1995. The use of habitat indices in water resource quality assessments. In *Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making*, W.S. Davis and T.P. Simon (eds.), pp. 181-208. Lewis Publishers, Boca Raton, FL.

Rapport, D. J., H. A. Regier, and T. C. Hutchinson. 1985. Ecosystem behavior under stress. *American Naturalist* 125:617-640.

Reeves, G. H., L. E. Benda, K. M. Burnett, P. A. Bisson, and J. R. Sedell. 1995. A disturbance-based ecosystem approach to maintaining and restoring freshwater habitats of evolutionarily significant units of anadromous salmonids in the Pacific Northwest. In *Evolution and the aquatic ecosystem: defining unique units in population conservation*, J. L. Nielsen (ed.), pp. 334-349. American Fisheries Society Symposium 17, Bethesda, Maryland, USA.

Regier, H.A., R.L. Welcomme, R.J. Steedman, and H.F. Henderson. 1989. Rehabilitation of degraded river ecosystems. In *Proceedings of the international large river symposium (LARS)*, D.P. Dodge (ed.), Canadian Special Publication of Fisheries and Aquatic Sciences 106, pp. 86-97. Department of Fisheries and Oceans, Ottawa, Ontario.

Reisner, M. 1986. *Cadillac desert: the American west and its disappearing water*. Penguin Books, New York.

Riebesell, J.F. 1974. Paradox of enrichment in competitive systems. *Ecology* 55: 183-187.

Richards, C., L.B. Johnson, and G.E. Host. 1996. Landscape-scale influences on stream habitats and biota. *Canadian Journal of Fisheries and Aquatic Sciences* 53:295-311.

Richards, C., L.B. Johnson. 1998. Landscape perspectives on ecological risk assessment. In *Risk Assessment: Logic and Measurement*, M.C. Newman and C. Strojan (eds). Ann Arbor Press.

Richardson, J. S., T. J. Lissimore, M. C. Healey, and T. G. Northcote. 2000. Fish communities of the lower Fraser River (Canada) and a 21-year contrast. *Environmental Biology of Fishes* 59:125-140.

Richter, B.D., D.P. Braun, M.A. Mendelson, and L.L. Master. 1997. Threats to imperiled freshwater fauna. *Conservation Biology* 11:1081-1093.

Rinne, J.N., R.M. Hughes, and B. Calamusso. 2005. *Historical changes in large river fish assemblages of the Americas*. American Fisheries Society, Symposium 45, Bethesda, Maryland.

Roni, P., T.J. Beechie, R.E. Bilby, F.E. Leonetti, M.M. Pollock and G.R. Pess. 2002. A review of stream restoration techniques and a hierarchical strategy for prioritizing restoration in Pacific Northwest watersheds. *North American Journal of Fisheries Management* 22:1-20.

Roth, N.E., J.D. Allan, D.L Erickson. 1996. Landscape influences on stream biotic integrity assessed at multiple spatial scales. *Landscape Ecology* 11:141-156.

Roy, A.H., A.D. Rosemond, M.J. Paul, D.S. Leigh, and J.B. Wallace. 2003. Stream macroinvertebrate response to catchment urbanisation (Georgia, USA). *Freshwater Biology* 48:329-346.

Schindler, D.W. 1987. Detecting ecosystem responses to anthropogenic stress. *Canadian Journal of Fisheries and Aquatic Sciences* 44:6-25.

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. *Environmental Monitoring and Assessment* 88: 119-152.

Shelton, A.D. and K.A. Blocksom. 2004. *A Review of Biological Assessment Tools and Biocriteria in New England States*. EPA/600/R-04/168. U.S. Environmental Protection Agency, Cincinnati, OH.

Simon, Thomas P. Ed. 2003. *Biological Response Signatures: Indicator patterns using aquatic communities*. CRC Press, Boca Raton, FL.

Simpson J.C and Norris R.H. 2000. Biological assessment of river quality: development of AusRivAS models and outputs. In *Assessing the Biological Quality of Fresh Waters: RIVPACS and Other Techniques*, J.F. Wright, D.W. Sutcliffe and M.T. Furse (eds.), pp. 125-142. Freshwater Biological Association, Ambleside, UK.

Slivitsky, M. 2001. *A Literature Review on Cumulative Ecological Impacts Of Water Use and Changes in Levels and Flows*. The Great Lakes Commission, October 15, 2001. 63 pp.

Smith, P.W. 1971. Illinois streams: a classification based on their fishes and an analysis of factors responsible for the disappearance of native species. *Illinois Natural History Survey Biological Notes* 76.

Smith, P.W. 1979. *The fishes of Illinois*. Univ. Illinois Press, Urbana, IL. 314 pp.

Smith, R.A., G.E. Schwartz and R.B. Alexander. 1997. Regional interpretation of water quality monitoring data. *Water Resources Research* 33: 2781-2798.

Smith, R.W., M. Bergen, S.B. Weisberg, D. Cadien, A. Dalkey, D. Montagne, J.K. Stull, and R.G. Velarde. 2001. Benthic response index for assessing infaunal communities on the southern California mainland shelf. *Ecological Applications* 11: 1073-1087.

Smith, S.H. 1972. The future of salmonid communities in the Laurentian Great Lakes. *Journal of the Fisheries Research Board of Canada* 29:951-957.

Snyder, C.D., J.A. Young, R. Villella, and D.P. Lemarie. 2003. Influences of upland and riparian land use patterns on stream biotic integrity. *Landscape Ecology* 18(7):647-664.

Solimini, A. G., G. A. Tarallo, and G. Carchini. 1997. Life history and species composition of the damselfly assemblage along the urban tract of a river in central Italy. *Hydrobiologia* 382:63-86.

Southerland, M.T. and J.B. Stribling. 1995. Status of biological criteria development and implementation. In *Biological assessment and criteria: tools for water resource planning and decision making*, W.S. Davis and T.P. Simon (eds), pp. 81-96. Lewis Publishers, Boca Raton, Florida.

Sparks, R.E. and W.C. Starrett. 1975. An electrofishing survey of the Illinois River, 1959-1974. *Illinois Natural History Survey Bulletin* 295-316.

State of Maine. 1985. Maine Laws Ch. 698 15 (in part). An Act to Amend the Classification System for Maine Waters and Change the Classifications of Certain Waters. Augusta, Maine.

State of Maine. 2003. Code of Maine Rules 06-096. Chapter 579: Classification Attainment Evaluation Using Biological Criteria for Rivers and Streams. Augusta, Maine.

State of Ohio. 2003. Ohio Administrative Code at OAC Chapter 3745-1. State of Ohio Water Quality Standards. Columbus, Ohio.

Steedman, R. J. 1988. Modification and assessment of an index of biotic integrity to quantify stream quality in southern Ontario. *Canadian Journal of Fisheries and Aquatic Sciences* 45:492-501.

Stevenson, R. J., Y. Pan, and P. Vaithiyanathan. 2002. Ecological assessment and indicator development in wetlands: the case of algae in the Everglades, USA. *Verhandlungen Internationale Vereinigung für Theoretische und Andgewandte Limnologie* 28:1248-1252.

Stoddard, J.L., D.P. Larsen, C.P. Hawkins, R.K. Johnson, and R.H. Norris. In Press. Setting expectations for the ecological condition of running waters: the concept of reference condition. *Ecological Applications*.

ten Brink, B.J.E., S.H. Hosper, and F. Colijn. 1991. A quantitative method for description and assessment of ecosystems: the AMOEBA approach. *Marine Pollution Bulletin* 23:265-270.

ten Brink, B.J.E. 1991. The AMOEBA approach as a useful tool for establishing sustainable development. In *In search of indicators of sustainable development*, O. Kuik and H. Vebruggen (eds.), pp. 71-87. Kluwer Academic Publishers, Dordrecht, The Netherlands.

Thoma, R.F. 1999. Biological monitoring and an index of biotic integrity for Lake Erie's nearshore waters. In *Assessing the Sustainability and Biological Integrity of Water Resources Using Fish Communities*, T.P. Simon (ed.), pp. 417-462. CRC Press, Boca Raton, FL.

Tockner, K. and J. A. Stanford. 2002. Riverine flood plains: present state and future trends. *Environmental Conservation* 29:308–330.

Trautman, M.B. 1957. *The Fishes of Ohio*. The Ohio State Univ. Press, Columbus, OH. 683 pp.

Trautman, M.B. 1981. *The Fishes of Ohio. Second edition*. Ohio State University Press, Columbus, OH.

Troelstrup, N. H. Jr., and G. L. Hergenrader. 1990. Effect of hydropower peaking flow fluctuations on community structure and feeding guilds of invertebrates colonizing artificial substrates in a large impounded river. *Hydrobiologia* 199:217-228.

Tsai, C. 1968. Effects of chlorinated sewage effluents on fishes in upper Patuxent River, Maryland. *Chesapeake Science* 9(2):83-93.

Tsai, C. 1973. Water quality and fish life below sewage outfalls. *Transactions of the American Fisheries Society* 102(2):281-292.

United States and Canada. 1987. Great Lakes Water Quality Agreement, as amended by protocol signed November 18, 1987. Windsor, ON.

U.S. Environmental Protection Agency (U.S. EPA). 1985. *Questions and Answers on Antidegradation*. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

U.S. Environmental Protection Agency (U.S. EPA). 2000a. *Mid-Atlantic Highlands streams assessment*. EPA/903/R-00/015. U.S. Environmental Protection Agency, Philadelphia, PA.

U.S. Environmental Protection Agency (U.S. EPA). 2000b. *Stressor Identification Guidance Document*. EAP/822/B-00/025. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

U.S. Environmental Protection Agency (U.S. EPA). 2002a. *Summary of Biological Assessment Programs and Biocriteria Development for States, Tribes, Territories, and Interstate Commissions: Streams and Wadeable Rivers*. EPA-822-R-02-048. U.S. Environmental Protection Agency, Office of Environmental Information and Office of Water, Washington, DC.

U.S. Environmental Protection Agency (U.S. EPA). 2002b. *National Water Quality Inventory 2000*. EPA-841-R-02-001. U.S. Environmental Protection Agency Office of Water, Washington DC 20460, August 2002.

U.S. Environmental Protection Agency (U.S. EPA). 2003. *Mid-Atlantic Integrated Assessment (MAIA): State of the flowing waters report*. U.S. Environmental Protection Agency, Corvallis, Oregon.

van Dam, H., A. Mertenes, and J. Sinkeldam. 1994. A coded checklist and ecological indicator values of freshwater diatoms from the Netherlands. *Netherlands Journal of Aquatic Ecology* 28:117-33.

Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell, and C.E. Cushing. 1980. The river continuum concept. *Canadian Journal of Fisheries and Aquatic Sciences* 37: 130-137.

Vansickle, J., and R.M. Hughes. 2000. Classification strengths of ecoregion, catchments and geographic clusters for aquatic vertebrates in Oregon. *Journal of the North American Benthological Society* 19:370-384.

Vitousek P.M., J.D. Aber, R.W. Howarth, G.E. Likens, P.A. Matson, D.W. Schindler, W.H. Schlesinger, and D.G. Tilman. 1997. Human alteration of the global nitrogen cycle: sources and consequences. *Ecological Applications* 7:737-750.

Vollenweider, R.A. 1968. *The scientific basis of lake and stream eutrophication with particular reference to phosphorus and nitrogen as eutrophication factors*. Technical report DAS/DSI/68.27. Organization for Economic Cooperation and Development, Paris.

Waite, I.R., and K.D. Carpenter. 2000. Associations among fish assemblage structure and environmental variables in Willamette basin streams, Oregon. *Transactions of the American Fisheries Society* 129:754-770.

Wallace, J.B., J.R. Webster, and W.R. Woodall. 1977. The role of filter feeders in flowing waters. *Archiv fur Hydrobiologie* 79:506-532.

Wallin, M., T. Wiederholm, and R.K. Johnson. 2003. *Guidance on establishing reference conditions and ecological status class boundaries for inland surface waters. European Union Common Implementation Strategy (CIS) for the Water Framework Directive.* Swedish University of Agricultural Sciences, Uppsala, Sweden.

Walsh, C.J., A.K. Sharp, P.F. Breen, and J.A. Sonneman. 2001. Effects of urbanisation on streams of the Melbourne region Victoria, Australia, I. Benthic macroinvertebrate communities. *Freshwater Biology* 46:553-565.

Wang, L., J. Lyons, P. Kanehl, and R. Gatti. 1997. Influences of Watershed Land Use on Habitat Quality and Biotic Integrity in Wisconsin Streams. *Fisheries* 22(6):6-12.

Wang, L., J. Lyons, P. Kanehl, R. Bannerman, and E. Emmons. 2000. Watershed Urbanization and Changes in Fish Communities in Southeastern Wisconsin Streams. *Journal of American Water Resources Association* 36(5):1173-1189.

Wang, L., J. Lyons, P. Kanehl, and R. Bannerman. 2001. Impacts of urbanization on stream habitat and fish across multiple spatial scales. *Environmental Management* 28:255-266.

Wang, L., J. Lyons, and P. Kanehl. 2002. Effects of watershed best management practices on habitat and fish in Wisconsin streams. *Journal of American Water Resources Association* 38:663-80.

Wang, L., P. Seelbach, and J. Lyons. In Press. Effects of levels of human disturbance on the influence of catchment, riparian, and reach scale factors on fish assemblages. In *Influences of landscapes on stream habitats and biological assemblages*, R. M. Hughes, L. Wang, and P. Seelbach (eds.), Pages xx-xx. American Fisheries Society Symposium xx. Bethesda, Maryland.

Ward, J.V. 1998. Riverine landscapes: biodiversity patterns, disturbance regimes, and aquatic conservation. *Biological Conservation* 83:269-278.

Welsh, H. H. Jr. and L. M. Ollivier. 1998. Stream amphibians as indicators of ecosystem stress: a case study from California's redwoods. *Ecological Applications* 8: 1118-1132.

Whittier, T. R., A. T. Herlihy, and S. M. Pierson. 1995. Regional susceptibility of Northeast lakes to zebra mussel invasion. *Fisheries* 20(6):20-27.

Whittier, T. R. and T. M. Kincaid. 1999. Introduced fish in Northeastern USA lakes: Regional extent, dominance and effect on native species richness. *Transactions of the American Fisheries Society* 128:769-783.

Whittier, T. R., J. L. Stoddard, R. M. Hughes, and G.A. Lomnicky. In Press. Associations among catchment- and site-scale disturbance indicators and biological assemblages at least- and most-disturbed stream and river sites in the western USA. In *Influences of landscapes on stream habitats and biological assemblages*, R. M. Hughes, L. Wang, and P. Seelbach (eds.), Pages xx-xx. American Fisheries Society Symposium xx. Bethesda, Maryland.

Wiens, J. A. 2002. Riverine landscapes: taking landscape ecology into the water. *Freshwater Biology* 47:501–515.

Wickham, J.D., K.B. Jones, K.H. Riitters, R.V. O'Neill, R.D. Tankersley, E.R. Smith, A.C. Neale and D.J. Chaloud. 1999. An integrated environmental assessment of the US Mid-Atlantic Region. *Environmental Management* 24:553–560.

Wilhelm, J.G.O., J.D. Allan, K.J. Wessell, R.W. Merritt and K.W. Cummins. Habitat evaluation of non-wadeable rivers. Unpublished manuscript.

Williams, J.E., C.A. Wood, and M.P. Dombeck (eds.). 1997. *Watershed restoration: principles and practices*. American Fisheries Society. Bethesda, Maryland.

Wissmar, R.C. and R.L. Beschta. 1998. Restoration and management of aquatic ecosystems: a catchment perspective. *Freshwater Biology* 40: 571-585.

Witte, F., T. Goldschmidt, J. Wanink, M. Van Oijen, K. Goudswaard, E. Witte-Maas, N. Bouton. 1992. The destruction of an endemic species flock: quantitative data on the decline of the haplochromine cichlids of Lake Victoria. *Environmental Biology of Fishes* 34:1-28.

Wright, J.F., D. Moss, P.D. Armitage, and M.T. Furse. 1984. A preliminary classification of running-water sites in Great Britain based on macro-invertebrate species and the prediction of community type using environmental data. *Freshwater Biology* 14: 221-256.

Wright, J.F., M.T. Furse, and P.D. Armitage. 1994. Use of macroinvertebrates communities to detect environmental stress in running waters. In *Water quality and stress indicators in marine and freshwater systems: linking levels of organisation*, D. W. Sutcliffe (ed.), pp. 15-34. Freshwater Biological Association, Ambleside.

Wright, J.F. 1995. Development of a system for predicting the macroinvertebrate fauna of flowing waters. *Australian Journal of Ecology* 20:181-197.

Wright, J.F. 2000. An introduction to RIVPACS. In *Assessing the Biological Quality of Fresh Waters*, J.F. Wright, D.W. Sutcliffe and M.T. Furse (eds.), pp. 1-24. Freshwater Biological Association, Ambleside, UK.

Wurtz, C.B. 1955. Stream biota and stream pollution. *Sewage & Industrial Wastes* 27(11):1270-1278.

Yang, J. R., F. R. Pick, and P. B. Hamilton. 1996. Changes in the planktonic diatom flora of a large mountain lake in response to fertilization. *Journal of Phycology* 32:232-243.

Yoder, C.O. 1978. *A proposal for the evaluation of water quality conditions in Ohio's rivers and streams*. Division of Industrial Wastewater, Columbus, OH. 43 pp.

Yoder, C.O. 1995. Policy issues and management applications for biological criteria. In *Biological assessment and criteria: tools for water resource planning and decision making*, W.S. Davis and T.P. Simon (eds.), pp. 327-343. Lewis Publishers, Boca Raton, FL.

Yoder, C.O. and E.T. Rankin. 1995a. Biological response signatures and the area of degradation value: New tools for interpreting multimetric data. In *Biological assessment and criteria: tools for water resource planning and decision making,* Davis, W.S. and Simon, T.P. (eds.), pp. 263-286. Lewis Publishers, Boca Raton, FL.

Yoder, C.O. and E.T. Rankin. 1995b. Biological criteria program development and implementation in Ohio. In *Biological assessment and criteria: tools for water resource planning and decision making*, Davis, W.S. and Simon, T.P. (eds.), pp. 109-144. Lewis Publishers, Boca Raton, FL.

Yoder, C.O. and E.T. Rankin. 1998. The role of biological indicators in a state water quality management process. *Environmental Monitoring and Assessment* 51:61-68.

Yoder, C.O. and E.T. Rankin. 1999. Biological criteria for water resource management. In *Measures of Environmental Performance and Ecosystem Condition*, P.C. Schulze and R.A. Frosch (eds.), pp. 227-259. National Academy of Engineering, National Academy Press, Washington, DC.

Yoder, C.O. and M. A Smith. 1999. Using fish assemblages in a state biological assessment and criteria program: essential concepts and considerations. In *Assessing the sustainability and biological integrity of water resources using fish communities*, T.P. Simon (ed.), pp. 17-56. CRC Press, Boca Raton, FL.

Yoder, C.O., R.J. Miltner, and D. White. 2000. Using biological criteria to assess and classify urban streams and develop improved landscape indicators. In *National Conference on Tools for Urban Water Resource Management and Protection*, S. Minamyer, J. Dye, and S. Wilson (eds.), pp. 32-44. EPA/625/R-00/001. U.S. EPA, Cincinnati, OH.

Yoder, C.O. and J.E. DeShon. 2003. Using biological response signatures within a framework of multiple indicators to assess and diagnose causes and sources of impairments to aquatic assemblages in selected Ohio rivers and streams. In *Biological response signatures: indicator patterns using aquatic communities*, T. P. Simon (ed.), pp. 23-81. CRC Press, Boca Raton, FL.

Yuan, L. L. and S. B. Norton. 2003. Comparing the responses of macroinvertebrate metrics to increasing stress. *Journal of the North American Benthological Society* 22:308-322.

Zaroban, D. W., M. P. Mulvey, T. R. Maret, R. M. Hughes, and G. D. Merritt. 1999. Classification of species attributes for Pacific Northwest freshwater fishes. *Northwest Science* 73:81-93.

Zeeb, B. A., C. E Christie, J. P. Smol, D. L. Findlay, H. J. Kling, and H. J. B. Birks. 1974. Responses of diatom and chrysophyte assemblages in lake-227 sediments to experimental eutrophication. *Canadian Journal of Fisheries and Aquatic Sciences* 51:2300-2311.

Glossary

Ambient Monitoring	sampling and evaluation of receiving waters not necessarily associated with episodic perturbations
Allochthonous	organic matter that was produced outside the system (e.g., wood, leaves, berries, insects etc.)
Anadromy	fish that live most of life in oceans or lakes and migrate to streams to spawn
Antidegradation Statement	statement that protects existing uses, prevents degradation of high quality waterbodies unless certain determinations are made, and which protects the quality of outstanding national resource waters
Aquatic Assemblage	an association of interacting populations of organisms in a given waterbody, for example, fish assemblage or a benthic macroinvertebrate assemblage
Aquatic Community	an association of interacting assemblages in a given waterbody, the biotic component of an ecosystem
Aquatic Life Use	a beneficial use designation in which the waterbody provides suitable habitat for survival and reproduction of desirable fish, shellfish, and other aquatic organisms; classifications specified in State water quality standards relating to the level of protection afforded to the resident biological community by the State agency
Attribute	measurable part or process of a biological system
Autochthonous	organic matter produced within the system (e.g., algae, macrophytes)
BEAST	used in parts of Canada, the BEAST (BEnthic Assessment of SedimenT) multivariate technique uses a probability model based on taxa ordination space and the "best fit" of the test site(s) to the probability ellipses constructed around the reference site classes
Beneficial Uses	desirable uses that water quality should support. Examples are drinking water supply, primary contact recreation (such as swimming), and aquatic life support.
Benthic Macroinvertebrates or Benthos	animals without backbones, living in or on the sediments, of a size large enough to be seen by the unaided eye and which can be retained by a U.S. Standard No. 30 sieve (28 meshes per inch, 0.595 mm openings). Also referred to as benthos, infauna, or macrobenthos
Best Management Practice	an engineered structure or management activity, or combination of these, that eliminates or reduces an adverse environmental effect of a pollutant
Biological Assessment or Bioassessment	an evaluation of the biological condition of a waterbody using surveys of the structure and function of a community of resident biota.
Biological Criteria or Biocriteria	**Scientific meaning**: quantified values representing the biological condition of a waterbody as measured by structure and function of the aquatic communities typically at reference condition. **Regulatory meaning**: narrative descriptions or numerical values of the structure and function of aquatic communities in a waterbody necessary to protect the designated aquatic life use, implemented in, or through water quality standards.

Biological Diversity or Biodiversity refers to the variety and variability among living organisms and the ecological complexes in which they occur. Diversity can be defined as the number of different items and their relative frequencies. For biological diversity, these items are organized at many levels, ranging from complete ecosystems to the biochemical structures that are the molecular basis of heredity. Thus, the term encompasses different ecosystems, species, and genes.

Biological Indicator or Bioindicator an organism, species, assemblage, or community characteristic of a particular habitat, or indicative of a particular set of environmental conditions

Biological Integrity the ability of an aquatic ecosystem to support and maintain a balanced, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitats within a region

Biological Monitoring or Biomonitoring use of a biological entity as a detector and its response as a measure to determine environmental conditions. Ambient biological surveys and toxicity tests are common biological monitoring methods.

Biological Survey or Biosurvey collecting, processing, and analyzing a representative portion of the resident aquatic community to determine its structural and/or functional characteristics

Bioregion any geographical region characterized by a distinctive flora and/or fauna

Clean Water Act an act passed by the U.S. Congress to control water pollution (formally referred to as the Federal Water Pollution Control Act of 1972). Public Law 92-500, as amended. 33 U.S.C. 1251 et seq.

Clean Water Act 303(d) This section of the Act requires States, territories, and authorized Tribes to develop lists of impaired waters for which applicable water quality standards are not being met, even after point sources of pollution have installed the minimum required levels of pollution control technology. The law requires that these jurisdictions establish priority rankings for waters on the lists and develop TMDLs for these waters. States, territories, and authorized Tribes are to submit their list of waters on April 1 in every even-numbered year.

Clean Water Act 305(b) biennial reporting requires description of the quality of the Nation's surface waters, evaluation of progress made in maintaining and restoring water quality, and description of the extent of remaining problems

Cosmopolitan Species species with worldwide distribution or influence where there is suitable habitat

Criteria limits on a particular pollutant or condition of a waterbody presumed to support or protect the designated use or uses of a waterbody. Criteria may be narrative or numeric.

DELT Anomalies percentage of Deformities, Erosions (e.g., fins, barbels), Lesions and Tumors on fish assemblages

Designated Uses those uses specified in water quality standards for each waterbody or segment whether or not they are being attained

Disturbance human activity that alters the natural state and can occur at or across many spatial and temporal scales

Ecological Integrity the condition of an unimpaired ecosystem as measured by combined chemical, physical (including physical habitat), and biological attributes. Ecosystems have integrity when they have their native components (plants, animals and other organisms) and processes (such as growth and reproduction) intact.

Ecoregion a relatively homogeneous ecological area defined by similarity of climate, landform, soil, potential natural vegetation, hydrology, or other ecologically relevant variables

Ecosystem-level functions — processes performed by ecosystems, including, among other things, primary and secondary production; respiration; nutrient cycling; decomposition. See discussion concerning how this function is considered in the draft biological condition gradient in transmittal memorandum under "outstanding issues" and in the file: attribute explanation.

Existing Uses — those uses actually attained in a waterbody on or after November 28, 1975, whether or not they are included in the water quality standards (November 28, 1975 is the date on which U.S. EPA promulgated its first water quality standards regulation). Because an existing use has been attained, it cannot be removed unless uses are added that require more stringent criteria.

Function — processes required for normal performance of a biological system (may be applied to any level of biological organization)

Heterotrophic — obtaining organic matter from other organisms rather than synthesizing it from inorganic substrates

Hyporheic Zone — area below the streambed where water percolates through spaces between the rocks and cobbles. Also known as the interface between surface water and groundwater.

Historical Data — data sets from previous studies, which can range from handwritten field notes to published journal articles

Historically documented taxa — taxa known to have been supported in a waterbody or region prior to enactment of the Clean Water Act, according to historical records compiled by state or federal agencies or published scientific literature

Index of Biological/Biotic Integrity — an integrative expression of site condition across multiple metrics. An index of biological integrity is often composed of at least seven metrics

Invasive species — a species whose presence in the environment causes economic or environmental harm or harm to human health. Native species or non-native species may show invasive traits, although this is rare for native species and relatively common for non-native species. (Please note - this term is not currently included in the biological condition gradient)

Life-history requirements — environmental conditions necessary for completing life cycles (including, among other things, reproduction, growth, maturation, migration, dispersal)

Lithophils — organisms that thrive on rocks or stones

Lithopelagophils — organisms that spawn in open gravelly areas and have no guarding behavior

Maintenance of populations — sustained population persistence; associated with locally successful reproduction and growth

Metric — a calculated term or enumeration representing some aspect of biological assemblage, function, or other measurable aspect and is a characteristic of the biota that changes in some predictable way with increased human influence

Multimetric Index — an index that combines indicators, or metrics, into a single index value. Each metric is tested and calibrated to a scale and transformed into a unitless score prior to being aggregated into a multimetric index. Both the index and metrics are useful in assessing and diagnosing ecological condition. See Index of Biotic Integrity.

Multivariate Analysis — statistical methods (e.g. ordination or discriminant analysis) for analyzing physical and biological community data using multiple variables

Narrative Biocriteria — written statements describing the structure and function of aquatic communities in a waterbody necessary to protect a designated aquatic life use

Native	an original or indigenous inhabitant of a region; naturally present
Non-detrimental effect	does not displace native taxa
Non-native or intentionally introduced species	with respect to a particular ecosystem, any species that is not found in that ecosystem. Species introduced or spread from one region of the U.S. to another outside their normal range are non-native or non-indigenous, as are species introduced from other continents.
Numeric Biocriteria	specific quantitative measures of the structure and function of aquatic communities in a waterbody necessary to protect a designated aquatic life use
Periphyton	a broad organismal assemblage composed of attached algae, bacteria, their secretions, associated detritus, and various species of microinvertebrates
Piscivore	predatory fish that eats mainly other fish
Polyphils	organism with no specialized spawning requirements, behavior, or preferred habitat
P/R	ratio of photosynthesis to respiration in a system
Presently Attained Uses	those uses actually being attained in a waterbody at the present moment
Rapid Bioassessment Protocols	cost-effective techniques used to survey and evaluate the aquatic community to detect aquatic life impairments and their relative severity
Reference Condition (Biological Integrity)	the condition that approximates natural, un-impacted conditions (biological, chemical, physical, etc.) for a waterbody. Reference condition (Biological Integrity) is best determined by collecting measurements at a number of sites in a similar waterbody class or region under undisturbed or minimally disturbed conditions (by human activity), if they exist. Since undisturbed or minimally disturbed conditions may be difficult or impossible to find, least disturbed conditions, combined with historical information, models or other methods may be used to approximate reference condition as long as the departure from natural or ideal is understood. Reference condition is used as a benchmark to determine how much other water bodies depart from this condition due to human disturbance.
Reference Condition (Biological Integrity), cont.	**Least Disturbed Condition:** the best available existing conditions with regard to physical, chemical, and biological characteristics or attributes of a waterbody within a class or region. These waters have the least amount of human disturbance in comparison to others within the waterbody class, region or basin. Least disturbed conditions can be readily found, but may depart significantly from natural, undisturbed conditions or minimally disturbed conditions. Least disturbed condition may change significantly over time as human disturbances change. **Minimally Disturbed Condition:** the physical, chemical, and biological conditions of a waterbody with very limited, or minimal, human disturbance in comparison to others within the waterbody class or region. Minimally disturbed conditions can change over time in response to natural processes. **Best Attainable Condition:** a condition that is equivalent to the ecological condition of (hypothetical) least disturbed sites where the best possible management practices are in use. This condition can be determined using techniques such as historical reconstruction, best ecological judgment and modeling, restoration experiments, or inference from data distributions

Reference Site	a site selected for comparison with sites being assessed. The type of sites selected and the type of comparative measures used will vary with the purpose of the comparisons. For the purposes of assessing the ecological condition of sites, a reference site is a specific locality on a waterbody that is undisturbed or minimally disturbed and is representative of the expected ecological integrity of other localities on the same waterbody or nearby waterbodies
Refugia	accessible microhabitats or regions within a stream reach or watershed where adequate conditions for organism survival are maintained during circumstances that threaten survival, e.g., drought, flood, temperature extremes, increased chemical stressors, habitat disturbance, etc.
Regional Reference Condition	a description of the chemical, physical, or biological condition based on an aggregation of data from reference sites that are representative of a waterbody type in an ecoregion, subecoregion, watershed, or political unit
Rheophils	organisms that flourish in free-flowing water
Restoration	the re-establishment of pre-disturbance aquatic functions and related physical, chemical, and biological characteristics
River Invertebrate Prediction and Classification System (RIVPACS)	a predictive method developed for use in the United Kingdom to assess water quality using a comparison of observed biological species distributions to those expected to occur based on a model derived from reference data
Sensitive taxa	intolerant to a given anthropogenic stress; first species affected by the specific stressor to which they are "sensitive" and the last to recover following restoration
Sensitive or regionally endemic taxa	taxa with restricted, geographically isolated distribution patterns (occurring only in a locale as opposed to a region), often due to unique life history requirements. May be long-lived, late maturing, low fecundity, limited mobility, or require mutualist relation with other species. May be among listed E/T or special concern species. Predictability of occurrence often low, therefore, requires documented observation. Recorded occurrence may be highly dependent on sample methods, site selection and level of effort.
Sensitive - rare taxa	naturally occur in low numbers relative to total population density but may make up large relative proportion of richness. May be ubiquitous in occurrence or may be restricted to certain micro-habitats, but because of low density, recorded occurrence is dependent on sample effort. Often stenothermic (having a narrow range of thermal tolerance) or cold-water obligates; commonly k-strategists (populations maintained at a fairly constant level; slower development; longer life-span). May have specialized food resource needs or feeding strategies. Generally intolerant to significant alteration of the physical or chemical environment; are often the first taxa observed to be lost from a community.
Sensitive - ubiquitous taxa	ordinarily common and abundant in natural communities when conventional sample methods are used. Often having a broader range of thermal tolerance than Sensitive- Rare taxa. These are taxa that comprise a substantial portion of natural communities, and that often exhibit negative response (loss of population, richness) at mild pollution loads or habitat alteration.
Spatial and temporal ecosystem connectance	access or linkage (in space/time) to materials, locations, and conditions required for maintenance of interacting populations of aquatic life; the opposite of fragmentation; necessary for metapopulation maintenance and natural flows of energy and nutrients across ecosystem boundaries
Stressors	physical, chemical, and biological factors that adversely affect aquatic organisms

Structure	taxonomic and quantitative attributes of an assemblage or community, including species richness and relative abundance structurally & functionally redundant attributes of the system = characteristics, qualities, or processes that are represented or performed by more than one entity in a biological system
Subcategorized Uses	States and Tribes may adopt subcategories of a use and set the appropriate criteria to reflect varying needs of such subcategories of uses, for instance, to differentiate between cold water and warm water fisheries
Taxa	a grouping of organisms given a formal taxonomic name such as species, genus, family, etc.
Taxa of intermediate tolerance	comprise a substantial portion of natural communities; may be r-strategists (early colonizers with rapid turn-over times; "boom/bust population characteristics). May be eurythermal (having a broad thermal tolerance range). May have generalist or facultative feeding strategies enabling utilization of relatively more diversified food types. Readily collected with conventional sample methods. May increase in number in waters with moderately increased organic resources and reduced competition but are intolerant of excessive pollution loads or habitat alteration.
Tolerant taxa	comprise a low proportion of natural communities. Taxa often are tolerant of a broader range of environmental conditions and are thus resistant to a variety of pollution or habitat induced stress. They may increase in number (sometimes greatly) in the absence of competition. Commonly r-strategists (early colonizers with rapid turn-over times; "boom/bust" population characteristics), able to capitalize when stress conditions occur. Last survivors.
Total Maximum Daily Load	the sum of the allowable loads of a single pollutant from all contributing point and nonpoint sources; calculation of the maximum amount of a pollutant a waterbody can receive and still meet water quality standards and an allocation of that amount to the pollutant's source
Use Attainability Analysis	structured scientific assessment of the physical, chemical, biological or economic factors affecting attainment of the uses of waterbodies
Water Quality Standards	a law or regulation that consists of the designated use or uses of a waterbody, the narrative or numerical water quality criteria (including biocriteria) that are necessary to protect the use or uses of that particular waterbody, and an antidegradation policy
Water Resource Management (Non-Regulatory)	decisions on management activities relevant to a water resource such as problem identification, need for and placement of best management practices, pollution abatement actions, and effectiveness of program activity

Acronyms

ACOE	Army Corps of Engineers
ALU	Aquatic Life Use
BCG	Biological Condition Gradient
BEAST	BEnthic Assessment of SedimenT
BOD	Biological Oxygen Demand
BPJ	Best Professional Judgment
BMP	Best Management Practice
CDG	Catchment Disturbance Gradient
CAFO	Confined Animal Feeding Operation
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CSO	Combined Sewer Overflows
CWA	Clean Water Act
EPT	Ephemeroptera, Plecoptera, Trichoptera
FWPCA	Federal Water Pollution Control Act
GIS	Geographic Information Systems
GLEI	Great Lakes Environmental Indicators
HDG	Human Disturbance Gradient
HDI	Human Disturbance Index
IBI	Index of Biological/Biotic Integrity
ICI	Invertebrate Community Index
ITFM	Intergovernmental Task Force on Monitoring Water Quality
LDM	Linear Discriminant Model
LWD	Large Woody Debris
NPDES	National Pollutant Discharge Elimination System
NWHI	Non-Wadeable Habitat Index
ONRW	Outstanding Natural Resource Waters
QHEI	Qualitative Habitat Evaluation Index
RBP	Rapid Bioassessment Protocols
RCRA	Resource Conservation and Recovery Act
RDG	Riparian Disturbance Gradient

RIVPACS	River Invertebrate Prediction and Classification System
TALU	Tiered Aquatic Life Use
TMDL	Total Maximum Daily Load
SCI	Stream Condition Index
STP	Sewage Treatment Plants
UAA	Use Attainability Analyses
WLAs	Waste Load Allocations
WQS	Water Quality Standards
WWTP	Wastewater Treatment Plant

Appendix A

MAINE TALU IMPLEMENTATION CASE HISTORY[1]

I. Establish conceptual foundation

Since the early 1970s, prior to adoption of the CWA, Maine water quality law has had a tiered structure, based on a gradient of water quality conditions. An early articulation of the conceptual basis for a tiered approach to establishing aquatic life uses was made by John Cairns and others in a U.S. EPA-sponsored symposium on the biological integrity objective of the Clean Water Act (Ballentine and Guarraia 1977), with further elaboration in Cairns et al. (1993) and Karr and Chu (2000). The underlying basis depicts biological condition declining across a gradient of stressors.

Maine's goal-based management classes range from Class AA, the highest water quality standard and greatest restrictions on human activity, to Class C (and formerly Class D, discontinued), the lowest quality standard with more flexible allowances for human activities (MDEP 2004 305b report). Maine's current water quality classification law for rivers and streams establishes four tiers of aquatic life use (ALU) that represent the upper end of a gradient of biological condition that occurs in the State (State of Maine 1985, Courtemanch et al. 1989, Courtemanch 1995). Conditions worse than this upper end (i.e., worse than Class AA/A, B, or C) are deemed unacceptable. Numeric biocriteria are based on assessment of benthic macroinvertebrates (State of Maine 2003, Davies et al. unpublished manuscript). Assessment of algal assemblages also occurs in most waterbodies but numeric criteria have not yet been developed. Maine relies on the response of benthic macroinvertebrates to human influences for several reasons:

- Diverse life history strategies and a wide range of pollution tolerance;
- Relatively long-lived (+/- 1 year) compared to algae and bacteria;
- Limited mobility diminishes stressor avoidance behavior and emigration;
- The indigenous fish assemblage in Maine is not very diverse and information is limited to just a few species.

Biologists in Maine and elsewhere have long observed clear-cut differences in community structure and composition of benthic macroinvertebrate samples that are collected from waters across a continuum of increasing stressors. The conceptual foundation of the Maine Department of Environmental Protection (MDEP) Biological Monitoring Program (and resulting biocriteria) was framed by three factors: 1) the first-hand observations of such biological response patterns, 2) published empirical and theoretical work in aquatic stress ecology, and 3) Maine's pre-existing water management context. The first two factors are discussed in sequence in this section. The water management context is discussed in the next section, II. *Merge Scientific & Policy Foundations*.

Empirical Observations of Maine Biologists

Differences in resident biological assemblages are evident even to the untrained eye when there are substantial differences in water quality (Figure A-1). This can be illustrated with a very simple example based on a gradient of increasing enrichment. In the initial years of biological assessment in Maine, biologists observed that minimally disturbed sampling locations tended to support many invertebrate taxa (high diversity), but at low to moderate density. In contrast, streams receiving well-treated or well-diluted domestic effluents exhibited higher organism densities, though the types of organisms were similar.

[1] Appendix A was written by Susan Davies, Maine Department of Environmental Protection.

Streams receiving heavy loadings of sewage or nutrient-laden industrial effluents showed obvious differences in taxa and numbers from that expected in minimally disturbed streams. Streams receiving toxic amounts of chlorine or industrial waste showed much lower densities and many more hardy types of organisms than would be expected in undisturbed areas.

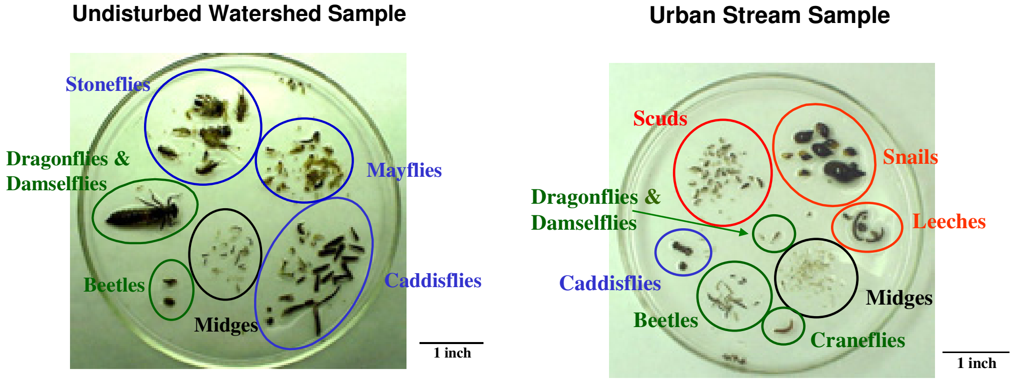


FIGURE A-1. Differences in numbers and types of organisms that are associated with different levels of disturbance can be evident even to the untrained eye.

Published Empirical and Theoretical Work in Aquatic Stress Ecology

The very obvious differences in biological responses for Maine streams, described above, are consistent with published conceptual models and empirical findings of stress ecology. The subsidy-stress gradient model of Reibesell (1974), and further developed by Odum et al. (1979) and Odum (1985), provided Maine DEP biologists with a theoretical model of expected patterns of biological change that was consistent with their own empirical observations (Figure A-2a and A-2b). Development of numeric biocriteria proceeded from this underlying ecological paradigm with the goal to statistically characterize the observed biological condition groups to determine aquatic life use class attainment.

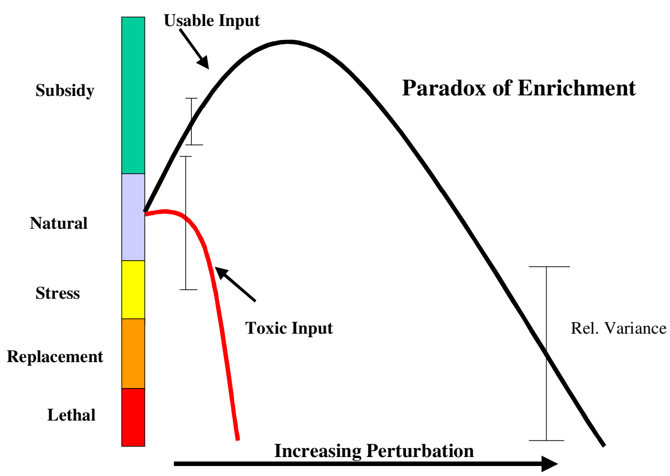


FIGURE A-2a. Subsidy-stress gradient: The ecological theory basis for Maine's aquatic life use descriptions (Odum et al. 1979). Some disturbances have an enriching or subsidizing effect on biological assemblages because they provide more than normal usable resources (nutrients, organic matter, etc.). Inputs in excess of what can be processed by the resident community have a detrimental effect (increased biochemical oxygen demand, accumulation of unusable resources, etc.) and lead to negative community response. Toxic or poisonous inputs have an immediate detrimental effect.

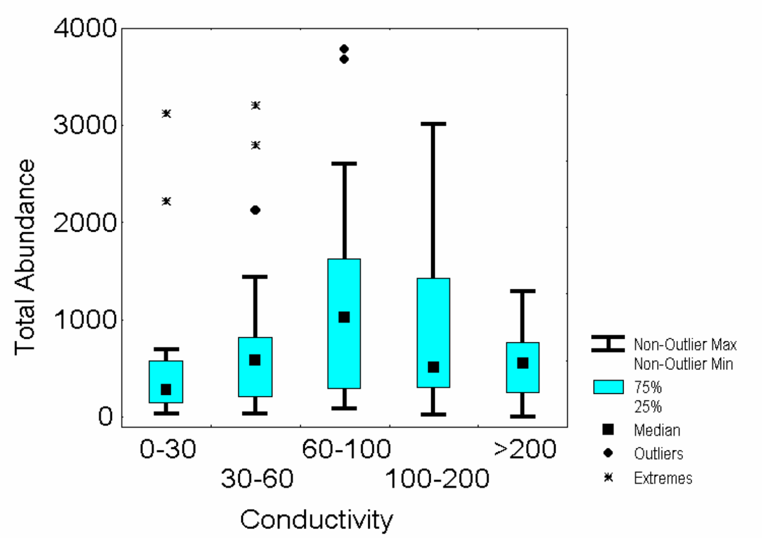


FIGURE A-2b. Empirically observed subsidy-stress gradient in Maine streams, documented by changes in benthic macroinvertebrate density. Low levels of conductivity are an indicator of slight enrichment while high levels are often associated with toxic contamination.

Stress ecology recognizes biological changes in response to increasing levels of stressors (i.e., gradients of environmental quality) as distinct from those that occur in responses to natural gradients, such as elevation, climate, alkalinity, stream size, and geographic location. While natural and ecoregional gradients can and do influence biological expectations in important ways, biological responses from the high to the low end of generalized stressor gradients in Maine streams tend to be far more obvious (Davies et al. 1999, Davies et al. unpublished manuscript). Odum's model supported our observation that structurally distinct biological groups exist across a gradient of water quality. Identifying predictable, characteristic differences among those biological condition groups could serve as the underlying conceptual basis for development of tiered aquatic life uses. Four biological condition groups would also fit well with the State's four-tiered standards for dissolved oxygen, bacteria, and habitat described in the existing water quality classification law.

II. Merge scientific and policy foundations

The narrative aquatic life use statements in Maine's TALUs describe conditions ranging from "as naturally occurs" (Class AA and Class A- the highest ALU designations) to "maintenance of structure and function" (Class C- the lowest ALU designation allowed in Maine) (Table A-1). The subsidy-stress gradient model helped guide the development of the ecologically-based definitions in the law. These specific definitions establish the biological characteristics that are required for attainment of each ALU classification (Table A-2).

TABLE A-1. Maine's narrative aquatic life and habitat standards for rivers and streams (M.R.S.A Title 38 Article 4-A § 464-465).

CLASS	MANAGEMENT	BIOLOGICAL STANDARD
AA*	High quality water for recreation and ecological interests. No discharges or impoundments permitted.	Habitat shall be characterized as natural and free flowing. Aquatic life shall be as naturally occurs.
A	High quality water with limited human interference. Discharges limited to non-contact process water or highly treated wastewater of quality equal to or better than the receiving water. Impoundments allowed.	Habitat shall be characterized as natural. Aquatic life shall be as naturally occurs
B	Good quality water. Discharge of well-treated effluent with ample dilution permitted. Impoundments allowed.	Habitat shall be characterized as unimpaired. Discharges shall not cause adverse impacts to aquatic life. Receiving water shall be of sufficient quality to support all aquatic species indigenous to the receiving water without detrimental changes in the resident biological community.
C	Acceptable water quality. Maintains the interim goals of the Federal Water Quality Act (fishable/swimmable). Discharge of well-treated effluent permitted. Impoundments allowed.	Habitat for fish and other aquatic life. Discharges may cause some changes to aquatic life, provided that the receiving waters shall be of sufficient quality to support all species of fish indigenous to the receiving water and maintain the structure and function of the resident biological community.
Impound-ments	Riverine impoundments not classified as Great Ponds and managed for hydropower generation	Support all species of fish indigenous to those waters and maintain the structure and function of the resident biological community.

*The narrative aquatic life standard is the same for Class AA and Class A.

TABLE A-2. Definitions of terms used in Maine's water classification law.

1. **Aquatic life** any plants or animals that live at least part of their life cycle in fresh water.
2. **As naturally occurs** conditions with essentially the same physical, chemical and biological characteristics as found in situations with similar habitats, free of measurable effects of human activity.
3. **Community function** mechanisms of uptake storage and transfer of life-sustaining materials available to a biological community, which determine the efficiency of use and the amount of export of the materials from the community.
4. **Community structure** the organization of a biological community based on numbers of individuals within different taxonomic groups and the proportion each taxonomic group represents of the total community.
5. **Indigenous** supported in a reach of water or known to have been supported according to historical records compiled by State and Federal agencies or published in scientific literature.
6. **Natural** living in or as if in, a state of nature not measurably affected by human activity.
7. **Resident biological community** aquatic life expected to exist in a habitat, which is free from the influence of the discharge of any pollutant. This shall be established by accepted biomonitoring techniques.
8. **Unimpaired** without a diminished capacity to support aquatic life.
9. **Without detrimental changes in the resident biological community** no significant loss of species or excessive dominance by any species or group of species attributable to human activity.

Consistency with other applicable WQ criteria

As shown in Figure A-3, MDEP designed the narrative ALUs to be parallel to the tiered dissolved oxygen and bacteria standards. This was done because Department biologists recognized that differences in allowed human activities and water quality criteria of the different classes (AA, A, B, C) would inevitably yield different expectations for aquatic community response. For example, it is unreasonable to expect the same biological assemblages to thrive in both Class AA waters (dissolved oxygen: "as naturally occurs"- >7 ppm for Maine; dams and discharges prohibited) and Class C waters (minimum dissolved oxygen 5 ppm; dams, industrial and municipal discharges allowed).

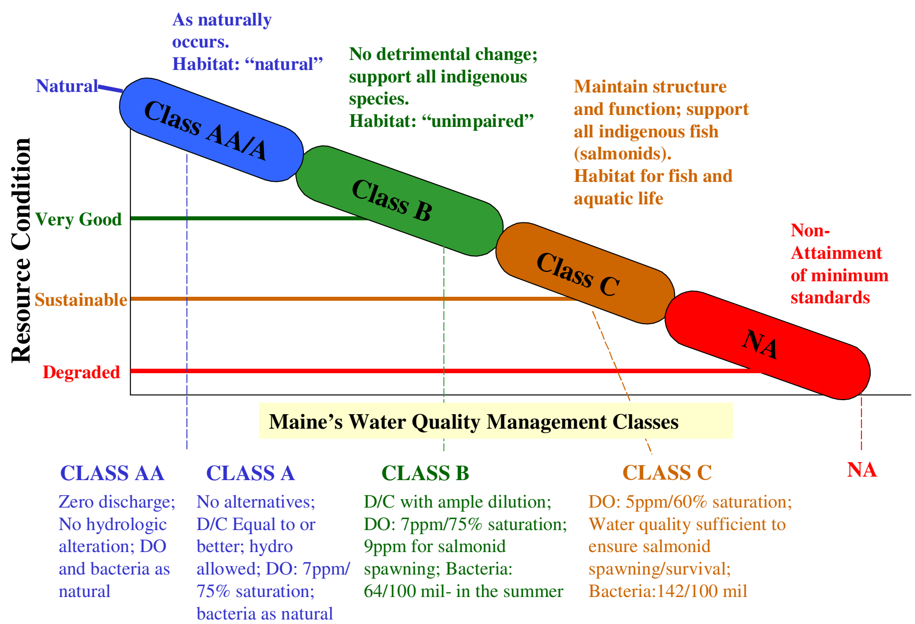


FIGURE A-3. Relation between Maine TALUs and other water quality standards and criteria.

The final language of the narrative aquatic life uses was the result of extensive negotiations between MDEP biologists and stakeholder biologists, under the purview of a legislative subcommittee. Lawyers on both sides weighed in regularly to ensure the fairness and legality of the statute. MDEP biologists drafted the narrative standards and definitions with careful attention to retaining a sound foundation in ecological theory. Furthermore, careful attention was given to how each biological attribute could be quantified (and thus assessed for attainment), with credible and widely accepted biological metrics (Table A-3).

TABLE A-3. Maine tiered uses based on measurable ecological values.

Narrative Standard	Ecological Value	Quantifiable Measures
CLASS A *natural*	Taxonomic and Numeric Equality; Presence of Indicator Taxa	Similarity, Richness, Abundance, Diversity; EPT, Indicator Taxa, Biotic Index
CLASS B *unimpaired, maintain indigenous taxa*	Retention of taxa and numbers; Absence of hyperdominance; Presence of sensitive taxa	Community loss; Richness; Abundance; Diversity; Equitability; Evenness; EPT; Indicator Taxa, Biotic Index
CLASS C *maintain structure*	Resistance, Redundancy; Resilience; Balanced Distribution	Richness; Diversity; Equitability; Evenness
and function	Energy exchange; Resource assimilation; Reproduction	Trophic groups; Richness; Abundance; Community loss; Fecundity; Colonization rate

How do Maine's tiered aquatic life uses relate to the Biological Condition Gradient?
Maine's aquatic life standards specify different levels (tiers) of water quality necessary to maintain designated aquatic life uses. These standards correspond to the tiers of the Biological Condition Gradient in Figure A-4.

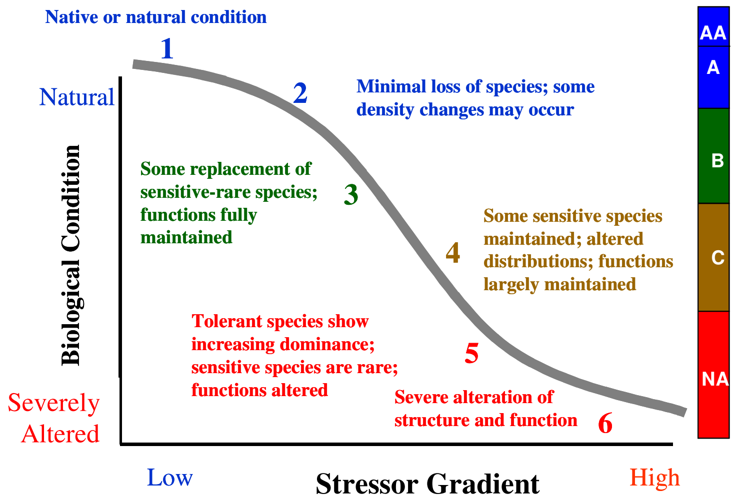


FIGURE A-4. Maine TALUs in relation to the BCG tiers.

<u>Class AA and Class A</u> have the same narrative aquatic life uses requiring that aquatic life be "as naturally occurs." This phrase is defined in the statute as "conditions with essentially the same physical, chemical, and biological characteristics as found in situations with similar habitats, free of measurable effects of human activity." The stated goal condition for Class AA/A thus conforms to Tier 1 or high Tier 2 conditions on the BCG.

Samples attaining MDEP Class A numeric criteria cover a range of conditions, some of which are fully consistent with BCG Tier 1 but some of which would have to be interpreted as BCG Tier 2. Examples of the latter are mildly enriched locations showing higher abundance of organisms (than "natural" for Maine) and increased algal biomass, and Class A locations that are influenced by dams.

<u>Class B</u> aquatic life standards require that there be "no adverse impacts" and that water quality be "sufficient to support all indigenous aquatic species without detrimental changes in the resident biological community." This phrase is defined as "no significant loss of species or excessive dominance by any species or group of species attributable to human activity." This wording was carefully chosen to allow for commonly observed increases in measures of biomass, density, and richness that occur in response to mild enrichment (as depicted by Odum's "subsidy hump" in Figure A-2a and A-2b) but to prohibit negative biological changes, such as notable loss of indigenous taxa. Thus the expectation for Class B is that sensitive taxa should be well represented with community structure comparable to Class A.

Samples attaining MDEP Class B numeric criteria cover a range of conditions, some of which are fully consistent with BCG Tier 2 but some of which would have to be interpreted as BCG Tier 3 because of the degree of structural change or the failure to collect Sensitive-Rare taxa. Dams, well-managed landscape changes, and well-treated point sources are allowed in Class B waters. These changes may result in detectable signals such as absence of migratory taxa, increased algal biomass, higher total abundance of organisms, and increased abundance of sensitive-ubiquitous taxa (i.e., higher relative abundance of some mayflies and some filter feeders; higher abundance of Perlid stoneflies) resulting in a community structure more consistent with Tier 3.

Class C aquatic life standards require that structure and function of the resident biological community be maintained. Numeric biocriteria in Maine document that waterbody segments meeting Class C dissolved oxygen and bacteria standards, but not attaining Class B standards, show obvious differences in biological assemblages. In terms of benthic macroinvertebrates, differences can be generally described as lower numbers and richness of cold-water obligate taxa and those taxa that have high dissolved oxygen requirements (e.g., gill-breathing mayflies and stoneflies), higher densities of filter-feeding organisms, and increased densities of some types of chironomid midges and other facultative or tolerant groups.

Samples attaining MDEP Class C numeric criteria cover a range of biological conditions, most of which are fully consistent with BCG Tier 3 and/or Tier 4. About 10% of samples that attain MDEP Class C numeric criteria would have to be interpreted as BCG Tier 5 because of the degree of structural change or very low numbers of Sensitive taxa (e.g., the mean abundance of Ephemeroptera in sites attaining Class C numeric criteria is 86 individuals per sampler but about 10% have less than 10 mayflies). Attainment of Class C numeric criteria usually indicates that other community structure attributes are present (e.g., evenness of distributions, richness and/or diversity of the assemblage of taxa of intermediate tolerance). Hyper-dominance of filter-feeders, complete absence of expected sensitive insect taxa (especially stoneflies and mayflies), and high proportions of tolerant taxa signal assemblages that fail to meet Class C water quality standards. These conditions represent BCG Tiers 5 and 6.

III. Establish technical program

How does Maine DEP collect biological data?

The MDEP's Biological Monitoring Program began standardized sampling of river and stream macroinvertebrates in 1983 (less rigorously standardized biological assessments had begun at least 10 years before). Experience gained on the Penobscot River (Davies 1987, Rabeni et al. 1988) had demonstrated the practical usefulness and reliability of rock-filled basket artificial substrates (Klemm et al. 1990). Maine has adapted the basic design of these devices to enable sampling of waterbody depths ranging from as little as 5 cm (using rock-filled mesh bags; Davies et al. 1999) to about 10 meters in large riverine impoundments (using boat-retrievable cones; Courtemanch 1984, Davies and Tsomides 2002, http://www.state.me.us/dep/blwq/docmonitoring/biological/biorep2000.htm). The success of these devices has enabled the MDEP to apply comparable field and analytical methods to nearly all rivers and streams of significant regulatory interest (Davies and Tsomides 2002), greatly simplifying the development and application of river and stream biocriteria. Further, the physiography of Maine is quite homogeneous with roughly 85% of the State falling within just two relatively similar ecoregions (Omernik 1987). For this reason stratification by ecoregion was not the critical concern that it is for States in some other regions of the country (Davies et al. unpublished manuscript).[2]

In 1999, Maine began an algal monitoring program to strengthen the interpretation of ecological condition by providing information from a second biological assemblage. Maine's fish assemblage is naturally depauperate, limiting its suitability as a candidate for bioassessment. The algal monitoring program will assist the Department in the development of river and stream nutrient criteria. The Department also has a companion biomonitoring program to assess wetland biological condition.

Database development

By the late summer of 2004, the Department had established about 800 monitoring stations in all major watersheds throughout the State (Figure A-5). Data from macroinvertebrate samples are stored in an Oracle© database and all stations are geo-referenced in the Department's geographic information system

---

[2]Maine's southern ecoregion is very small but recent data suggest that some improvement in accuracy of class prediction could result from better accounting for ecoregional differences there.

(ArcInfo©). Data collected in accordance with Maine's biocriteria protocol are analyzed using statistical models that estimate to which of the four water quality classes a sample belongs. Findings of the Biological Monitoring Program are used to document existing conditions, identify problems, set water management goals, assess the progress of water resource management measures, and trigger needed remedial actions.

Sampling methods
Samples of benthic macroinvertebrates are collected from flowing streams in rock bags (or baskets or cones). At least three substrate samplers are exposed in the waterbody for 28 days during the late summer, low flow period (July 1 to September 30). The MDEP usually conducts sampling, but others may also perform monitoring to determine attainment of classification if done according to a quality assurance plan.

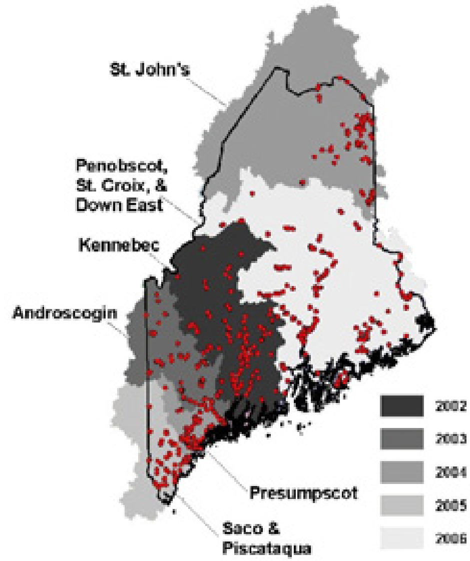


FIGURE A-5. Macroinvertebrate sampling stations in Maine.

Laboratory methods
Samples are retrieved, sorted, and stored for identification by a professional freshwater macroinvertebrate taxonomist. Organisms are identified to species whenever possible or otherwise to the lowest taxonomic level possible.

Analytical methods
If a sample satisfies the minimum data requirements (total mean abundance of at least 50 individuals, generic richness of at least 15 taxa for 3 replicate samplers), data are entered into the MDEP's computer software for further analysis through the numeric criteria statistical model. The model is able to take large amounts of information generated from a biological sample, describe which variables appear to be most significant in the classification decisions, and provide a mathematical summary that integrates the information. The model produces probability scores from 0 to 1 that indicate the likelihood that a sample attains each water quality class.

IV. Develop and validate quantitative thresholds

How does Maine quantify the tiered aquatic life uses so that attainment can be assessed?
In the late 1980's, the MDEP quantified the narrative aquatic life goals for each water quality class by developing a probability-based statistical model to serve as numeric biocriteria (Courtemanch et al. 1989, Courtemanch 1995, Davies et al. unpublished manuscript). The model uses 31 biological variables, many of which were specifically chosen because of their utility in measuring some important ecological attribute in the narrative standard. The model quantifies and standardizes the expert judgment of biologists and it now serves as an expert system for decision-making (*See Case Examples 3-3 and 3-6*).

To develop the model, biologists used agreed-upon decision rules and a Delphi technique (Bakus et al. 1982) to assign an aquatic life attainment classification (A, B, C, or non-attainment) to 144 samples of benthic macroinvertebrate data, based on conformity of the sampled community to one of the 3 narrative aquatic life standards in Maine's statute, or to a fourth category representing non-attainment of minimum State standards (Shelton and Blocksom 2004, Davies et al. unpublished manuscript). The samples evaluated represented 300 distinct taxonomic units and 70,000 organisms collected from rivers, streams, and riverine impoundments. Those data and their classification assignments were used as the baseline for construction of the expert system, in the form of a linear discriminant model, to evaluate future macroinvertebrate samples for water quality classification attainment. The original model was used from 1992 through 1999 when the model was recalibrated with an additional 229 (for a total of 373) sampling events. The recalibration resulted in relatively minor changes to the structure of the original model, involving simplification of the structure of two of the sub-models, the elimination of two poorly performing variables, and changes in model coefficients to account for the new data.

How has Maine established reference conditions?

Maine has taken a conceptually different approach to establishing baseline reference conditions from which to develop numeric biological criteria. Because we determined that detection of four distinct biological condition groups, characterized by differences in specified ecological attributes, was our management goal, it was also our goal for statistical analysis. We desired to develop numeric criteria that would enable us to assign sites to one of those four condition groups (A, B, C, non-attainment). Therefore, our task for characterizing reference conditions was to conceptually and then statistically define those four groups. Thus in a sense, initially by expert judgment and then by multivariate analysis, we created a Class A reference condition (deemed to be close to natural), a Class B reference condition, a Class C reference condition, and non-attainment reference conditions. Use of biological information to establish a minimally disturbed reference has been criticized due to the dangers of a too circular process. We have tested our biology-based a priori assignment of sites to Class A using more traditionally identified reference locations (i.e., based on high percent natural landcover) and found good correspondence with the biologically-defined Class A sites.

Adoption of the Numeric Biocriteria Rule

On April 17, 2003 the Maine Board of Environmental Protection adopted numeric freshwater biocriteria in rule. The biocriteria rule describes the process that the MDEP uses to make decisions about attainment of aquatic life uses in rivers and streams. The rule describes protocols for biological sampling of benthic macroinvertebrates, laboratory analyses, modeling analysis of laboratory data, and selective use of expert judgment. Adoption of this rule quantitatively interprets Maine's existing narrative 'aquatic life' standards for each riverine water quality classification.

V. Application in water quality management

How does the MDEP decide which waterbodies and locations to monitor?

For purposes of biological monitoring, the MDEP divided the State into five major river basins, which are sampled on a 5-year rotational schedule (Figure A-6): Androscoggin, Kennebec and Mid-Coast, Penobscot, St. Croix and North Coastal Rivers, Piscataqua, Saco and Southern Coast, St. John and Presumpscot. The decision to monitor specific locations on a waterbody can be based on a variety of factors such as:

- prior knowledge of human activities that could have a detrimental effect on a waterbody: sampling seeks to detect actual impacts on biological communities;
- knowledge of future potential threats to a waterbody: sampling can be done to collect baseline data before, for example, development occurs or a discharge is licensed; follow-up sampling can determine the effect, if any, on the biological community by said development or discharge;
- requirement/desire to monitor the effects of remediation activities or water quality management changes;
- desire to expand coverage of the monitoring program and to more fully document natural variability.

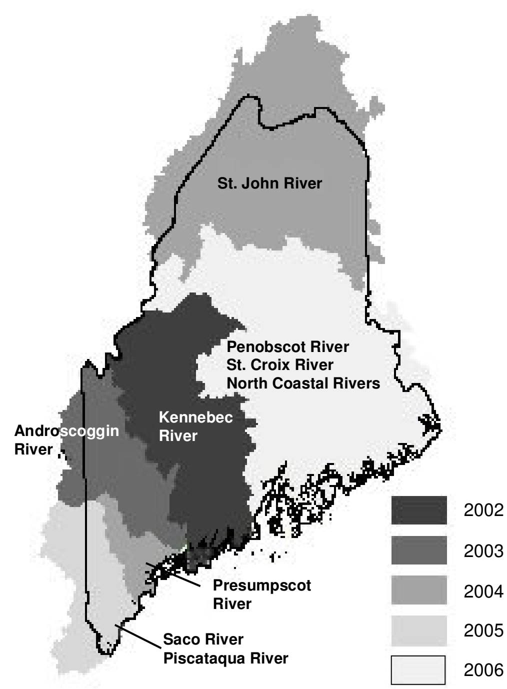


FIGURE A-6. Maine five-year rotating basin sampling schedule.

How are tiered aquatic life uses designated in Maine?
The quality of Maine's waters is described in terms of physical, chemical and biological characteristics associated with the State's water classification program. As established in Maine statute (38 MRSA Sections 464-470), the classification program consists of designated uses (e.g. drinking water supply, recreation in and on the water, habitat for fish and other aquatic life), criteria (e.g. bacteria, dissolved oxygen and aquatic life), and characteristics (e.g. natural, free flowing) that specify levels of water quality necessary to maintain the designated uses. All State waters have a classification assignment (Rivers and streams: AA, A, B, C; Lakes: GPA; Marine and estuarine: SA, SB, SC). Tiered narrative aquatic life uses specific to wetlands are currently under consideration by MDEP and a supporting wetland biomonitoring program is in place.

The classification system in Maine is goal-based in that assignment of a given waterbody to a use class (AA, A, B or C) may not necessarily reflect its current conditions. Rather, it establishes the level of quality the State has deemed the waterbody must achieve. Maine's classification system is also more risk based than quality based. Water quality differences among the various classes are not large, however, the different levels of restrictions put on human activities associated with each class establishes the level of risks that water quality could be degraded resulting in increased threats to designated use attainment. Rivers and streams are assigned to a tiered aquatic life use goal (Table A-1: AA and A -"*as naturally occurs,*" B- "*no detrimental change,*" C- "*maintain structure and function and water quality sufficient to support salmonids*") that represents the best fit after considering:

- The current condition in terms of dissolved oxygen, bacteria, and aquatic life (Figure A-3) and
- The highest attainable goal condition (taking into account ecological and socioeconomic factors).

The State water quality assessment provided in Maine's 305b report gives the status of attainment of the water resource goals established in the classification program. Thus, some waters may be listed as impaired even though they have relatively good water quality (Table A-4), e.g., a Class A river may be listed because it does not fully attain the standards of that class but may be of sufficiently good quality to attain Class B or C, and the Clean Water Act interim goal. The classification program is reviewed every three years (Triennial Review) by the Department and the Board of Environmental Protection (Board). The Board may, after opportunity for public review and hearing, make recommendations to the Legislature for changes in water quality standards or reclassification of selected waters. The most recent revisions to the classification program were completed in 2002-2003 when the Legislature authorized classification upgrades to 75 river, stream and coastal segments totaling over 800 miles of waters (Figure A-7).

TABLE A-4. Examples of how numeric biocriteria results determine whether or not a waterbody attains designated aquatic life uses in Maine.

Legislative Class	Monitoring Result	Attains Class?	Next Step
A	A	Yes	--
C	B	Yes	--
A	B	No	TMDL
B	NA	No	TMDL

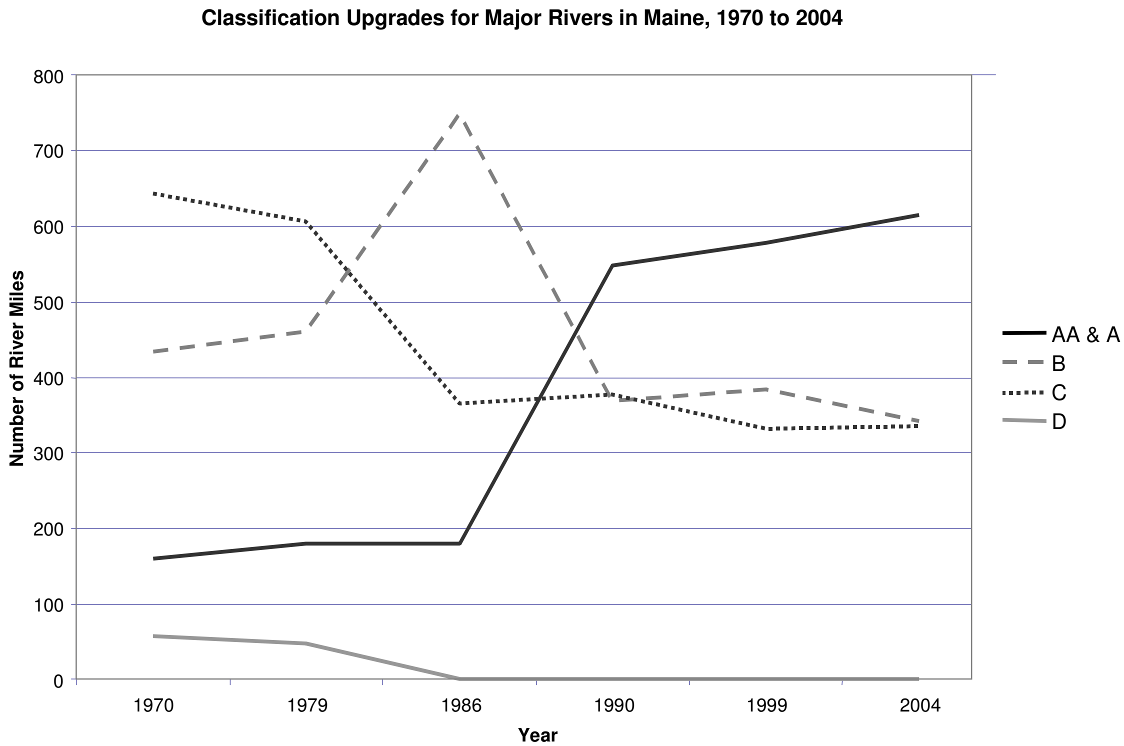


FIGURE A-7. Increased designation of Class AA and Class A uses on major Maine rivers (as shown by river miles) between 1970 and 2004, as a result of water quality improvements and public support for the Class AA/A goal in the Triennial Review Process.

What is the management perspective for TALU designations in Maine?

Class AA waterbodies, as compared to Class A, have significantly greater restrictions on allowed activities. For example, no discharge of wastewater and no dams are allowed in Class AA waterbodies. Class A waters carry a higher risk of degradation because discharges are allowed, though the risk is small because they must be of "equal to or better" water quality than the receiving water. Dams are also allowed. Obstructions to flow, whether man-made or natural can alter assemblage structure from free-flowing conditions (Poff et al. 1997, Davies et al. 1999). The definition in water quality standards for the term "natural" sought to limit the effects of altered flows to no greater than what could be expected from a "natural" obstruction to flow (e.g., a natural hydrological control or a beaver dam). Thus to accommodate dams in Class A, "*natural*" is defined as "occurring in, *or as if in*, a state of nature not measurably affected by human activities." Assemblages that are characteristic of the waters above and below beaver dams or low-head, run-of-river, man-made dams are deemed to pass this standard. Most dams in Class A provide for passage of anadromous fish.

Class B was originally applied as the default ALU for unmonitored waters though current use designations are nearly equal in stream miles for Class A and Class B, both of which far exceed Class C miles when all rivers and streams in the State are considered (Figure A-8). From the management perspective, a Class B designation often applies to waterbody segments exposed to well-treated or well-diluted domestic discharges or to areas subjected to landscape alterations that result in moderate increases in the nutrient and organic matter load.

Class C narrative aquatic life standards prohibit any activities that result in the loss of structure and function of the resident biological community. "Community structure" is defined as "the organization of a biological community based on numbers of individuals within different taxonomic groups and the proportion each taxonomic group represents of the total community," while community function is defined as "mechanisms of uptake storage and transfer of life-sustaining materials available to a biological community which determine the efficiency of use and the amount of export of the materials from the community." This management class is applied to waterbodies that may be impounded, altered by landscape changes, or that receive industrial wastewater.

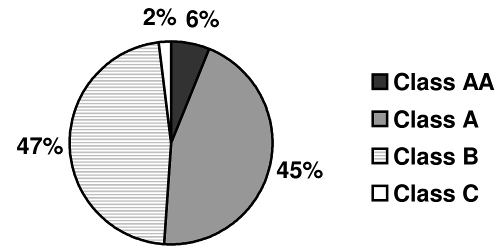


FIGURE A-8. Percent of linear miles of all rivers and streams in each of Maine's designated use classes (year 2000).

What process was used to bring the Maine TALU biocriteria rule through adoption?

The MDEP Biological Monitoring Program completed provisional numeric biocriteria in 1990. Those numeric thresholds were the basis for extensive regulatory and non-regulatory Department decisions between 1990 and 2003, e.g., issuance or denial of 401 water quality certificates and recommendations for flow management changes, 303d and 305b listings, prioritization of at-risk waterbodies, and problem identification. In April 2003, the State formally adopted tiered numeric biocriteria rules that were the result of the analysis of 15 years of biological data and the experience gained through 20 years of regulatory decision-making based on numeric biocriteria (Table A-5). Remarkably, the biocriteria rule was one of the most complicated and important, but least contested water quality rules that the Maine Department of Environmental Protection has adopted in the last 15 years. Stakeholders from all sides had become convinced of the merits of the approach.

TABLE A-5. Chronology of Maine's biocriteria development.

1983	The MDEP Biological Monitoring Program began a standardized program of sampling stream invertebrate communities.
1986	The revised Water Classification Program, which defined tiered narrative standards for aquatic life, became law.
1989	MDEP staff and University of Maine statistical ecologist, Dr. Frank Drummond embarked on the development of numeric criteria to support the narrative standards of the law.
1990	A technical advisory committee of stakeholder scientists was convened to provide peer review and oversight of the biocriteria development process. Over the course of approximately 2 ½ years, MDEP staff, Dr. Drummond, and the committee developed a statistical model based on expert judgment and linear discriminant analysis to address the scientific goals, as well as the policy and regulatory goals of the new biocriteria program.
1991-1993	Public informational workshops on the process were held in March 1991, September 1993, and December 1993.
1999	The original statistical model was recalibrated to take advantage of the expanded dataset available at that time.
2002	During a formal stakeholder review process, meetings were held in March and April and comments were solicited from representatives of the hydropower and paper industry, environmental advocacy groups, other State agency biologists (e.g., fish and wildlife), university scientists, and private consultants.
2002	A workshop on the rule and its background was held in early October for the Maine Board of Environmental Protection.
2003	The Board of Environmental Protection adopted the rule on April 3 and it was subsequently adopted by the Maine State Legislature.

Appendix B

OHIO TALU IMPLEMENTATION CASE HISTORY[1]

In 1990, Ohio EPA adopted numeric biological criteria in the Ohio Water Quality Standards (Ohio WQS; Ohio Administrative Code 3745-1). These criteria have been used to guide and enhance water quality management programs and assess their environmental outcomes. The numeric biocriteria are an outgrowth of an existing framework of tiered aquatic life uses and narrative biological assessment criteria that has been in place since 1980. This case history is intended to summarize the evolutionary development of the components of the WQS and monitoring and assessment programs that took place in the late 1970s and throughout the 1980s and 1990s.

I. Establish conceptual foundation

Initially developed and adopted by Ohio EPA in 1978, tiered aquatic life uses represented a major revision to the existing general use framework that was adopted in 1974. This level of tiered uses recognized the different types of warmwater aquatic assemblages that corresponded to the mosaic of natural features of the landscape and nearly two centuries of human-induced changes. The eventual development of more refined tiered uses and the attendant numeric biocriteria that are in place today was the result of a decade long development process. The important concepts that spurred and guided these developments in the Ohio EPA program are described as follows:

Natural History and Zoogeography
The empirical evidence used to develop the initial concepts for tiered uses can be found in comprehensive works on the natural history and zoogeography of the Midwest such as Fishes of Ohio (Trautman 1957, 1981) and Fishes of Illinois (Smith 1979). These texts documented the natural and human-induced variations in the distribution, composition, and abundance of biological assemblages over space and through time. Trautman (1957) not only provides a lesson in Ohio's natural history, but also describes the biological evidence that was used to formulate the initial concepts about biological integrity that emerged in the late 1970s and early 1980s. Such works also described the key features of the landscape that influence and determine the potential aquatic fauna of waterbodies and were the forerunners of the regionalization tools that appeared soon after. As an alternative to a "one-size-fits-all" approach, these provided an important foundation for the development of Ohio's tiered uses.

Landmark Stream and River Pollution Studies
The earliest studies of the effects of pollution on biological assemblages were the precursors of the approach eventually developed and used by Ohio EPA. Campbell (1939), Brinely (1942), and Wurtz (1955) described the classical zones of pollution in flowing waterbodies. Ellis (1937) conducted one of the first comprehensive studies of water pollution in the U.S. including an emphasis on the chronic impacts of wastewater discharges. Patrick (1950, 1953) employed the concept of species (or taxa) diversity as an indicator of the "health and well-being" of aquatic assemblages and described a "biodynamic cycle." Gaufin and Tarzwell (1953) also described pollutional zones using aquatic assemblages and were the first to advocate cost-effective assessments of one or two representative assemblages (e.g., fish and macroinvertebrates). Subsequent studies of that time included landmark pollution investigations of rivers and streams (Krumholz and Minckley 1964; Mills et al. 1966; Tsai 1968, 1973; Sparks and Starrett 1975; Gammon 1976), some of which introduced standardized approaches to

[1] Appendix B was written by Chris Yoder, Midwest Biodiversity Institute, Columbus, Ohio.

biological data collection and analysis. These were the key citations in the original proposal for the present-day Ohio EPA biological assessment program (Yoder 1978). Such works also provided the impetus for articulating the linkage between ecological symptoms of aquatic health and human-induced changes in aquatic ecosystem quality that came later.

Concepts of Biological Integrity

The articulation of a practical definition of biological integrity by Karr and Dudley (1981) provided a theoretical framework for the development of Ohio's numeric biological criteria. Key components of this framework are: 1) using biological assemblages as a direct measure of aquatic life use attainment status (Herricks and Schaeffer 1985, Karr et al. 1986), 2) the development and use of multimetric assessment tools (Karr 1981, Karr et al. 1986), 3) derivation of regional reference condition to determine appropriate aquatic life use goals and assessment endpoints (Hughes et al. 1986), and 4) systematic monitoring and assessment of the State's waters. This represented a major advancement over previous attempts to define and develop a workable framework to address the concept of integrity (Ballentine and Guarraia 1977). Embedded in this framework is the recognition that water quality management must be approached from an ecological perspective that is grounded in sound ecological theory *and* validated by empirical observation. This means developing monitoring and assessment and WQS to encompass the five factors that determine the integrity of a water resource (Figure 1-3; Karr et al. 1986).

Experiences in Applying Systematic Biological Assessments

A major aspect of the development of the Ohio biological assessment program and tiered uses is the experience gained through the initial and sustained development of systematic bioassessments beginning in the late 1970s and through the 1980s. This is where the previously described methods, concepts, and theories were applied, tested, and developed, resulting in a tractable system for measuring biological quality at multiple spatial scales and through time. An evolutionary process occurred in which qualitative, narrative biocriteria were initially used to assess rivers and streams via systematic watershed monitoring and assessments. The data and experiences gained in this process provided the raw materials for incorporating the concepts of biological integrity that emerged simultaneously. This resulted in further refinements to the biological assessment tools and criteria and the tiered uses including how they are assigned and assessed. Key to the success of this approach was the initial decisions about indicator assemblages and methods. These have remained stable throughout the entire development and implementation process, with no major modifications that would have resulted in major disconnections of the database. The specific methods, tools, and criteria are described in Section II.

When numeric biocriteria and refined uses were adopted in 1990, the development process continued with adaptations of that system to different waterbody types. A systematic process for classifying and assessing wetlands was developed in the early 1990s and narrative biocriteria were adopted in the Ohio WQS. Biological assessment methods and indexes were also developed for the Lake Erie near shore and lacustuary habitats (Thoma 1999). Routine application of the numeric biocriteria in support of dredge and fill permitting and 401 certifications exposed the need to develop new assessment tools for primary headwater streams, i.e., those draining less than one square mile. Dealing with these waters required a change in indicator groups emphasizing aquatic amphibians and invertebrates and a modified classification scheme (Ohio EPA 2003). Finally, the Ohio River Valley Water Sanitation Commission (ORSANCO) developed a systematic approach for assessing fish (Emery et al. 2003) and macroinvertebrate assemblages of the Ohio River mainstem as a precursor to the adoption of numeric biocriteria. Other innovations are expected to follow and include recalibration of the stream and river biocriteria following the resampling of reference sites that took place during 1990-1999, urban stream classification issues (Yoder et al. 2000, Miltner et al. 2003), and adaptation to level IV ecoregions and other geomorphic classification schemes. These are examples of a continuous improvement process that naturally follows the adherence to the fundamentals of integrating WQS with systematic monitoring and assessment.

II. Merge scientific and policy foundations

From the outset, biological and water quality assessments were intended to play a pivotal role in the application of tiered uses. Since designated uses were formulated and described in ecological terms, it followed that they should be applied and measured on an ecological basis. At that time, the readily available criteria were chemical-specific and the development of practical and systematic biological assessments was in pilot testing and development stages. The operational execution of tiered uses (WQS) was dependent on developing a more comprehensive and systematic approach to monitoring and assessment that supported the watershed and waterbody specific application of tiered uses. However, time was required to develop standardized data, tools and criteria, spatial design, and spatial coverage, which were part of the monitoring and assessment program that delivered full support for tiered uses (and all other water quality management programs). Figure B-1 illustrates the evolutionary and incremental process of the development of tiered uses, allied tools and criteria, and the monitoring and assessment approach that were necessary to achieve full implementation of TALU in Ohio.

• General ALU • Few Specific Chemical Criteria • Narrative "Free Froms" • "Pilot" biological monitoring program • Fixed station M&A design (chemical only; 100+ sites)	• Narrative TALUs • More Specific Chemical Criteria • Initial designation of specific waters (BPJ based; system-atic M&A envisioned) • Fixed station M&A design (chemical only; 100+ sites) • "Pilot" biological monitoring program (10-15 sites/yr.)	• Narrative TALUs • More Specific Chemical Criteria • Narrative Biocriteria (initial WQ program support) • Designation of specific waters based on M&A (UAA process) • Intensive river & watershed surveys initiated (integration of biol, chem, and physical indicators; 100 -200 sites/yr.)	• Refined TALUs • Specific & Complex Chemical Criteria • Numerical Biocriteria (BCG implicit) • Physical Habitat Assessment • WET Testing • Geometric watershed design (integration of biol, chem, and physical indicators; 400-600 sites/yr.) • Integrated Bioassessments (systematic WQ program support)
(1974 - 1978)	(1978 - 1980)	(1980 - 1987)	(1987 - present)

FIGURE B-1. Evolutionary development of TALU and allied tools, criteria and assessments from the baseline of the 1974 WQS based on general uses and few specific water quality criteria to refined TALUs and specific chemical, physical, and biological criteria implemented via an integrated monitoring and assessment framework. The three time periods beginning with 1978-1980 approximate the first three phases of biocriteria development and implementation in Figure 5-2.

Pre-development Phase: 1974-1978

The first WQS adopted in 1974 were consistent with the technology available at that time consisted of general uses, "free from" statements, and few numeric criteria of any kind (chemical, physical, or biological). The monitoring and assessment program adhered to contemporary U.S. EPA guidance, consisting of a fixed station network (approximately 100 sites, monthly and quarterly chemical sampling) and a "pilot" biological program. The baseline water quality management programs (i.e., NPDES permitting, funding, planning) were also in their initial stages of development and implementation. A comprehensive water quality based approach to pollution abatement and management had not yet been developed or envisioned – abatement efforts focused on technology based limitations for major point sources. The linkage between WQS and monitoring and assessment had not yet been made, the latter being viewed as a less important, optional activity.

Initial TALU Development Phase: 1978-1980

In 1978, tiered aquatic life and other uses (e.g., recreation, water supply) were described and adopted along with the development of numeric chemical criteria for parameters such as dissolved oxygen (D.O.), temperature, ammonia, and common heavy metals (e.g., copper, cadmium, lead, zinc, iron, chromium, and nickel). The tiered uses emanated from recognition of the broader ecological concepts described in section I, as well as the belief that a "one-size-fits-all" approach to water quality management (i.e., the result of applying general uses) was neither realistic, cost-effective, nor saleable to stakeholders and the public. While tiered uses promised more customized and cost-effective management outcomes, the integration of WQS and monitoring and assessment, which is necessary before these stated objectives could be realized, had not yet taken place.

Ohio's First Tiered Use Designations

Tiered aquatic life uses are articulated as narrative statements describing the ecological attributes that should be supported by each tier. The criteria associated with each tier consisted of pollutant-specific, single value criteria for a limited set of water quality parameters (i.e., D.O., temperature, ammonia, common heavy metals). There were no biological criteria at that time, although the vision was to eventually develop a biologically-based assessment process. The tiers included variations on a theme of warmwater aquatic assemblages as written in the narrative for the warmwater habitat (WWH) use designation:

> "These are waters capable of supporting reproducing populations of fish, normally referred to as warmwater species, and associated vertebrate and invertebrate organisms and plants on an annual basis. These standards apply outside of the mixing zone." (Ohio Administrative Code 3745-1-07 c. 1978)

The intent of the exceptional warmwater habitat (EWH) use designation is illustrated by the phrase "These are waters capable of supporting ***exceptional and unusual*** populations of fish . . . " In essence, the EWH designation required evidence of an exceptional or unusual assemblage of fish or associated aquatic organisms and plants on an annual basis. Initially, EWH designations were made based on the known locations of self-sustaining populations of fish and other aquatic species that were considered of exceptional value, most of which had exhibited historical declines in distribution throughout Ohio and the Midwest in response to human-induced changes. These locations also corresponded to a congruence of natural landscape features associated with Ohio's glacial geology that "insulated" these assemblages from the cascade of effects from alteration in the landscape that adversely impacted the same species in other more vulnerable waterbodies. The result was waters with more intact habitats, less altered hydrological characteristics, and water quality that was "much better than most." As such, a goal of EWH is to protect such aquatic habitats as a refuge for rare and sensitive species and is vital to the broader restoration goals of the 1972 Federal Water Pollution Control Act (FWPCA) amendments. A greater degree of protection was initially afforded to these waters via more stringent water quality criteria for key parameters such as D.O., ammonia, and temperature (Ohio Administrative Code 3745-1-07 c. 1978). WWH became the default designation for all other waters that lacked such "exceptional and unusual attributes", but which retained or had the potential to exhibit the minimum quality that met the baseline provisions of the FWPCA (Sec. 101[a][2]).

A coldwater habitat (CWH) designation was also developed, but primarily focused on fishery attributes (i.e., Salmonids), which are largely artificially propagated and maintained in Ohio. However, the possibility of incorporating broader ecological attributes into this use narrative was included in the designated use narrative as follows:

> "These are waters capable of supporting populations of fish, normally referred to as coldwater species and associated vertebrate and invertebrate organisms and plants on an annual basis. These waters are not necessarily capable of supporting successful reproduction of Salmonids and may be stocked periodically. These standards apply outside of the mixing zone." (OAC 3745-1-07)

The monitoring and assessment program was initially based on fixed stations and emphasized chemical assessments, but experimental approaches such as small-scale intensive surveys and biological assessments were being developed and tested. There were no empirically derived or narrative biological criteria to decide between EWH and WWH. Specific assignments of waters were made using expert consensus and best professional judgment based on the known ecological attributes inherent in each designation. Thus the assignments of individual water bodies were only as good as the information available for such waters, which was later found to be incomplete or inadequate. Other tiers in the Ohio aquatic life use designations included seasonal warmwater habitat (SWH) and limited warmwater habitat (LWH). Water quality criteria for common chemical parameters were tiered and/or varied for each use designation. Criteria were the most stringent for CWH and EWH and the least stringent for LWH, the latter use essentially functioning as a temporary variance to WWH.

Initial TALU Implementation and Development Phase: 1980-1987

While the tiering provided by EWH and WWH is conceptually consistent with the intent and attributes of the biological condition gradient (BCG; Chapters 2 and 3), the tools to quantify and implement the associated concepts were lacking in 1978. The inclusion of the concepts of biological integrity (Karr and Dudley 1981), operational measures of biological condition (Karr et al. 1986), and the concepts of regionalization and reference sites (Hughes et al. 1986, Omernik 1987) led to further refinements of the tiered uses in this phase. These refinements resulted in the present day hierarchy of the exceptional warmwater, warmwater, modified warmwater, and limited resource waters use designations. The narrative descriptions were modified to reflect the operational definition of biological integrity (Karr and Dudley 1981), further integrating the parallel development of numeric biological criteria.

The original tiered uses were devised with an eye toward the eventual development of a biological assessment based approach to their implementation. These initial developments took place in the early 1980s and included narrative (or qualitative) biological "criteria" (Tables B-1 and B-2) supported by biological assessments and the implementation of an intensive survey design executed on a mainstem river or watershed basis (Ohio EPA 1981). These early biocriteria were based on the experiences and best professional judgment of the agency biologists and reflected the analytical and assessment tools of that time. At the same time, t chemical criteria were being further developed and whole effluent toxicity (WET) testing was being explored.

The use of monitoring and assessment in support of water quality management programs emphasized WQS (assigning tiered uses), construction grants (advanced treatment justifications), and NPDES permits (water quality based effluent limits). At the same time, the statewide database that would support the eventual and more comprehensive development of biological, chemical, and physical assessment tools and criteria was being amassed via the systematic implementation of a an intensive survey and watershed assessment process. Comparatively complex chemical-specific criteria were adopted for 126 priority pollutants and included chronic, acute, and lethal endpoints for aquatic life; criteria were also adopted for human health exposures. Whole effluent toxicity testing was introduced and developed as a water quality based permitting tool (Figure B-1).

TABLE B-1. Biological criteria (fish) for determining aquatic life use designations and attainment of Clean Water Act goals (November, 1980; after Ohio EPA 1981).

Evaluation Class Category	"Exceptional" Class I (EWH)	"Good" Class II (WWH)	"Fair" Class III	"Poor" Class IV
1.	Exceptional or unusual assemblage of species	Usual association of expected species	Some expected species absent, or in very low abundance	Most expected species absent
2.	Sensitive species abundant	Sensitive species present	Sensitive species absent, or in very low abundance	Sensitive species absent
3.	Exceptionally high diversity	High diversity	Declining diversity	Low diversity
4.	Composite index >9.0 – 9.5	Composite index >7.0 – 7.5; <9.0 – 9.5	Composite index >4.5 – 5.0; <7.0 – 7.5	Composite index <4.0 – 4.5
5.	Outstanding recreational Fishery		Tolerant species increasing, beginning to dominate	Tolerant species dominate
6.	Rare, endangered, or threatened species present			

Conditions: Categories 1, 2, 3, and 4 (if data is available) must be met and 5 or 6 must also be met in order to designated in a particular class.

TABLE B-2. Biological criteria (macroinvertebrates) for determining aquatic life use designations and attainment of Clean Water Act goals (November, 1980; after Ohio EPA 1981).

Evaluation Class Category	"Exceptional" Class I (EWH)	"Good" Class II (WWH)	"Fair" Class III	"Poor" Class IV
1.	Pollution sensitive species abundant	Pollution sensitive species present in moderate numbers	Pollution sensitive species present in low numbers	Pollution sensitive species absent
2.	Intermediate species present in low numbers	Intermediate species present in moderate numbers	Intermediate species abundant	Intermediate species present in low numbers or absent
3.	Tolerant species present in low numbers	Tolerant species present in low numbers	Tolerant species present in moderate numbers	Tolerant species abundant (all types may be absent if extreme toxic conditions exist)
4.	Number of taxa >30[1]	Number of taxa 25-30	Number of taxa 20-25	Number of taxa <20
5.	Exceptional diversity Shannon index <3.5	High diversity Shannon index 2.9-3.5	Moderate diversity Shannon index 2.3-2.9	Low diversity Shannon index <2.3

[1]Number of quantitative taxa from artificial substrates

A key development that took place during this time period was the pilot testing of ecoregions (Omernik 1987) and the development of the regional reference condition concept (Hughes et al. 1986). Along with the emerging concepts of biological integrity (Gakstatter et al. 1981, Karr and Dudley 1981) and multimetric assessment tools (Karr 1981, Karr et al. 1986), these advances represent the foundational development of the tools and criteria that emerged out of this phase. During this phase, integrated biological, chemical, and physical assessments were emphasized in support of a wider array of management issues (including nonpoint sources) in addition to the mainstay priorities of construction grants and NPDES permitting. The results of these assessments were documented in Comprehensive Water Quality Reports, the production of which included the first true integration of the monitoring and assessment, WQS, water quality modeling, and permitting programs. Study teams were formed for each project and included staff membership from each program. The analyses and recommendations included in these reports provided the basis for WQS use revisions, water quality based NPDES permits (including water quality certifications), advanced treatment justifications, and other findings related to the observed impacts of nonpoint sources.

The WQS were modified in 1985 to include a listing of designations by individual waterbody, as opposed to default designations or tributary membership (Table B-3). The original listing of individual waterbodies in the WQS was based on the Gazetteer of Ohio Rivers and Streams (Ohio Dept. of Natural Resources 1960). Waterbodies listed in the Gazetteer that had not been assessed via the biological and water quality assessment process were assigned a “default” designation of WWH. Waterbodies that were originally designated in 1978, or subsequent to that version of the WQS, retained those uses and this was denoted for each waterbody in the rules (Table B-3). Unconfirmed non-WWH uses required validation by site-specific monitoring and assessment due to a public notice issued by Ohio EPA in 1981. In reality, many “default” WWH designations also required reassessment because the variations in watershed settings and stressor gradients had only begun to be recognized. The Gazetteer of Ohio Rivers and Streams did not include all jurisdictional streams in the State; thus “unlisted” streams were assigned use designations as they became known via the systematic assessment of Ohio watersheds and/or as site-specific management issues arose. This further emphasized the role of monitoring and assessment in the designation of individual waterbodies.

Ongoing TALU Implementation and Maintenance Phase: 1987- present

Prompted by the testing and developments that took place in the initial implementation and development phase, Ohio EPA proposed and adopted numerical biological criteria (Figure B-2) and further refinements to the tiered uses. The narratives of the tiered uses first developed in 1978 were revised and new uses were added, both of which were influenced by the developments and the monitoring and assessment experience that took place in the preceding time period. The aquatic life use narratives were revised to reflect the operational definition of biological integrity (Karr and Dudley 1981) and provided direct reference to how the numerical biological criteria were developed and derived. These definitions follow:

> “Warmwater” – these are waters capable of supporting and maintaining a balanced, integrated, adaptive community of warmwater aquatic organisms having a species composition, diversity, and functional organization comparable to the twenty-fifth percentile of the identified reference sites within each of the following ecoregions: the interior plateau ecoregion, the Erie/Ontario lake plains ecoregion, the western Allegheny plateau ecoregion and the eastern corn belt plains ecoregion. For the Huron/Erie lake plains ecoregion, the comparable species composition, diversity and functional organization are based on the ninetieth percentile of all sites within the ecoregion. For all ecoregions, the attributes of species composition, diversity, and functional organization will be measured using the index of biotic integrity, the modified index of well-being, and the invertebrate community index as defined in “Biological Criteria for the Protection of Aquatic Life: Volume II, Users Manual for Biological Field Assessment of Ohio

Surface Waters," as cited in paragraph (B) of rule 3745-1-03 of the Administrative Code. In addition to those water body segments designated in rules 3745-1-08 to 3745-1-32 of the Administrative Code, all upground storage reservoirs are designated warmwater habitats. Attainment of this use designation (except for upground storage reservoirs) is based on the criteria in Table 7-14 of this rule. A temporary variance to the criteria associated with this use designation may be granted as described in paragraph (F) of rule 3745-1-01 of the Administrative Code.

TABLE B-3. Example of individual stream and/or segment use designations in the Ohio water quality standards showing aquatic life, water supply, and recreational use designations. Designation with a "+" means the use has been confirmed by monitoring and assessment. Designation with an "*" indicates a "default" designation or unverified designation – these waters will eventually be assessed via the rotating basin approach [excerpted from Ohio Administrative Code 3745-1-09].

Waterbody Segment	Use Designations													Comments
		Aquatic Life Habitat						Water Supply			Recreation			
	SRW	WWH	EWH	MWH	SSH	CWH	LRW	PWS	AWS	IWS	BW	PCR	SCR	
↓ ↓ ↓ ↓ ↓ ↓ ↓ ↓														
Scioto River – Frank Rd. (RM 127.7) to downstream from Bridge St. in Chillicothe (RM 70.7)		+							+	+		+		
- Greenlawn Dam (RM 129.8) to Frank Rd. (RM 127.7)	+	+							+	+		+		
- Olentangy R. (RM 132.3) to Greenlawn Dam (RM 129.8)	+			+					+	+		+		ECBP ecoregion – impounded MWH
- Dublin Rd. WTP dam (RM 133.4) to Olentangy R. (RM 132.3)	+	+							+	+		+		
- O'Shaughnessy Dam (RM 148.8) To Dublin Rd. WTP dam (RM 133.4)	+	+						+	+	+		+		
- all other segments		*							*	*		*		
Scippo Cr.			+						+	+		+		
Congo Cr. (Scippo Cr. at RM 1.64)			+						+	+		+		
Unnamed trib. Scippo Cr. (RM 16.31)		+							+	+			+	
Unnamed trib. Scippo Cr. (RM 18.87)							+		+	+			+	Small drainageway maintenance
Yellowbud Cr. - Ebenhack Rd. (RM 3.0) to mouth			+						+	+		+		
- all other segments		+							+	+		+		
RCA Tributary (Scioto R. RM 96.5)							+		+	+			+	Small drainageway maintenance

SRW = State Resource Water; WWH = Warmwater Habitat; EWH = Exceptional Warmwater Habitat; MWH = Modified Warmwater Habitat; SSH = Seasonal Salmonid Habitat; CWH = Coldwater Habitat; LRW = Limited Resource Waters; PWS = Public Water Supply; AWS = Agricultural Water Supply; IWS = Industrial Water Supply; BW = Bathing Waters; PCR = Primary Contact Recreation; SCR = Secondary Contact Recreation

Huron Erie Lake Plain (HELP)

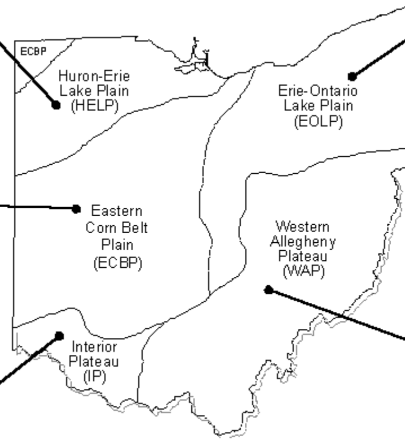
Use	Size	IBI	MIwb	ICI
WWH	H	28	NA	34
	W	32	7.3	34
	B	34	8.6	34
MWH-C	H	20	NA	22
	W	22	5.6	22
	B	20	5.7	22
MWH-I	B	30	5.7	NA

Erie Ontario Lake Plain (EOLP)

Use	Size	IBI	MIwb	ICI
WWH	H	40	NA	34
	W	38	7.9	34
	B	40	8.7	34
MWH-C	H	24	NA	22
	W	24	6.2	22
	B	24	5.8	22
MWH-I	B	30	6.6	NA

Eastern Corn Belt Plains (ECBP)

Use	Size	IBI	MIwb	ICI
WWH	H	40	NA	36
	W	40	8.3	36
	B	42	8.5	36
MWH-C	H	24	NA	22
	W	24	6.2	22
	B	24	5.8	22
MWH-I	B	30	6.6	NA



Western Allegheny Plateau (WAP)

Use	Size	IBI	MIwb	ICI
WWH	H	44	NA	34
	W	44	8.4	34
	B	40	8.6	34
MWH-C	H	24	NA	22
	W	24	6.2	22
	B	24	5.8	22
MWH-A	H	24	NA	30
	W	24	5.5	30
	B	24	5.5	30
MWH-I	B	30	6.6	NA

Interior Plateau (IP)

Use	Size	IBI	MIwb	ICI
WWH	H	40	NA	30
	W	40	8.1	30
	B	38	8.7	30
MWH-C	H	24	NA	22
	W	24	6.2	22
	B	24	5.8	22
MWH-I	B	30	6.6	NA

Statewide Exceptional Criteria

Use	.Size	IBI	MIwb	ICI
EWH	H	50	NA	46
	W	50	9.4	46
	B	48	9.6	46

FIGURE B-2. Numeric biological criteria adopted by Ohio EPA in 1990, showing stratification of biocriteria by biological assemblage, index, site type, ecoregion for the warmwater habitat (WWH) and exceptional warmwater habitat (EWH) use designations.

The narrative for the exceptional warmwater habitat (EWH) use designation retained the same application language with the following differences (in bold italics):

> ***"Exceptional warmwater"*** - these are waters capable of supporting and maintaining ***an exceptional or unusual*** community of warmwater aquatic organisms having a species composition, diversity, and functional organization comparable to the ***seventy-fifth percentile of the identified reference sites on a statewide basis*** . . . ***all lakes and reservoirs, except upground storage reservoirs, are designated exceptional warmwater habitats.*** Attainment of this use designation ***(except for lakes and reservoirs)*** is based on the criteria in Table 7-14 of this rule."

The narrative for coldwater habitat (CWH) was also revised and reflected a broader application of this use for reasons other than the existence of maintenance stocking of Salmonid fish species:

(i) "Coldwater habitat, inland trout streams" – these are waters which support trout stocking and management under the auspices of the Ohio department of natural resources, division of wildlife, excluding waters in lake run stocking programs, lake or reservoir stocking programs, experimental or trial stocking programs, and put and take programs on waters without, or without the potential restoration of, natural coldwater attributes of temperature and flow. The director shall designate these waters in consultation with the Director of the Ohio department of natural resources.

(ii) "Coldwater habitat, native fauna" – these are waters capable of supporting populations of native coldwater fish and associated vertebrate and invertebrate

> organisms and plants on an annual basis. The director shall designate these waters based upon the result of use attainability analyses.

The WWH, EWH, and CWH use designations are considered consistent with the minimum goals of the CWA (Section 101[a][2]) and the associated Federal Regulation (40CFR Part 130). However, the public notice issued in 1981 by Ohio EPA required that designated uses other than WWH be validated on a waterbody specific basis prior to basing permitting requirements on the attendant water quality criteria. Furthermore, a waterbody must reflect the capability to attain the EWH biological criteria at a sufficient number of sampling locations to be designated EWH (Ohio EPA 1987) and the CWH designation has its own set of requirements in the narrative. Such showings are not required for WWH, except that the potential to attain must be determined by biological and habitat assessments.

> "Coldwater" – these are waters that meet one or both of the characteristics described in paragraphs (B)(1)(f)(i) and (B)(1)(f)(ii) of this rule. A temporary variance to the criteria

Use designations that do not meet the minimum goals of the CWA, and thus require a use attainability analysis on a water body specific and/or segment-by-segment basis include:

> "Modified warmwater" – these are waters that have been the subject of a use attainability analysis and have been found to be incapable of supporting and maintaining a balanced, integrated, adaptive community of warmwater aquatic organisms due to irretrievable modifications of the physical habitat. Such modifications are of a long-lasting duration (i.e., twenty years and longer) and may include the following examples: extensive stream channel modification activities permitted under sections 401 and 404 of the act or Chapter 6131 of the Revised Code, extensive sedimentation resulting from abandoned mine land runoff, and extensive, permanent impoundment of free-flowing water bodies. The attributes of species composition, diversity and functional organization will be measured using the index of biotic integrity, the modified index of well-being, and the invertebrate community index as defined in "Biological Criteria for the Protection of Aquatic Life: Volume II, Users Manual for Biological Field Assessment of Ohio Surface Waters," as cited in paragraph (B) of rule 3745-1-03 of the Administrative Code. Attainment of this use designation is based on the criteria in Table 7-14 of this rule. The modified warmwater habitat designation can be applied only to those waters that do not attain the warmwater habitat biological criteria in Table 7-14 of this rule because of irretrievable modifications of the physical habitat. All water body segments designated modified warmwater habitat will be reviewed on a triennial basis (or sooner) to determine whether the use designation should be changed. A temporary variance to the criteria associated with this use designation may be granted as described in paragraph (F) of rule 3745-1-01 of the Administrative Code.

The Limited Resource Waters (LRW) use designation is defined as:

> "Limited resource water – these are waters that have been the subject of a use attainability analysis and have been found to lack the potential for any resemblance of any other aquatic life habitat as determined by the biological criteria in Table 7-14 of this rule. The use attainability analysis must demonstrate that the extant fauna is substantially degraded and that the potential for recovery of the fauna to the level characteristic of any other aquatic life habitat is realistically precluded due to natural background conditions or irretrievable human-induced conditions. All water body segments designated limited resource water will be reviewed on a triennial basis (or sooner) to determine whether the use designation should be changed. Limited resource waters are also termed nuisance

prevention for some water bodies designated in rules 3745-1-08 to 3745-1-30 of the Administrative Code. A temporary variance to the criteria associated with this use designation may be granted as described in paragraph (F) of rule 3745-1-01 of the Administrative Code. Waters designated limited resource water will be assigned one or more of the following causative factors. These causative factors will be listed as comments in rules 3745-1-08 to 3745-1-30 of the Administrative Code.

(i) "Acid mine drainage" – these are surface waters with sustained pH values below 4.1 s.u. or with intermittently acidic conditions combined with severe streambed siltation, and have a demonstrated biological performance below that of the modified warmwater habitat biological criteria.

(ii) "Small drainageway maintenance" – these are highly modified surface water drainageways (Usually less than three square miles in drainage area) that do not possess the stream morphology and habitat characteristics necessary to support any other aquatic life habitat use. The potential for habitat improvements must be precluded due to regular stream channel maintenance required for drainage purposes.

(iii) Other specified conditions.

The designation of specific waterbodies as MWH or LRW requires a use attainability analysis (UAA) based on a waterbody specific assessment. These do not meet the minimum conditions prescribed by the CWA (Section 101[a][2]). All of these were adopted in the Ohio WQS in 1990.

Relationship of Ohio's Tiered Uses to the Biological Condition Gradient

Ohio's current tiered uses represent refinements to the original tiered uses adopted in 1978 and reflect the developments that benefited from ten years of experience in applying a tiered use system. The practical impacts of these refined and tiered uses on water quality management are described in Table B-4 and include the designated use, the key attributes of that use, why a waterbody would be designated for that use, and some of the practical impacts to water quality management. All of the biological criteria and some of the chemical/physical criteria associated with each use are tiered in a logical relationship to the ecological attributes, which are ascribed by the designated use narrative and the translation of that narrative to specific criteria. This is consistent with the concepts of the BCG in that expectations and attainment of each use are measured by the biological criteria that are in turn designed to describe and measure increments in quality along the BCG (Figure B-3). Chemical-specific and physical parameters are cast in the role of stressor and exposure indicators and criteria (i.e., they are best used as design criteria in modeling and TMDLs). They directly support the development and implementation of abatement and management strategies via water quality management programs by providing the translation between associations of cause and effect via monitoring and assessment to enforceable controls via permitting and best management practices via TMDLs. The biological criteria are cast in the role of response indicators and as the primary criteria for determining use attainment status, measuring relative quality, and documenting the effectiveness of abatement and management strategies (Yoder and Rankin 1998, Karr and Yoder 2004). The logical relationship between exposure and response follows in that some of the key chemical criteria are more stringent for the uses that are representative of the higher tiers of the BCG (i.e., EWH) and least stringent for the lowest tiers (i.e., MWH, LRW). These are then translated accordingly to wastewater and other water quality management requirements. However, criteria that do not demonstrate an empirical relationship along the BCG are not tiered.

TABLE B-4. Key features associated with tiered aquatic life uses in the Ohio WQS (OAC 3745-1-07).

Aquatic Life Use	Key Attributes	Why a Waterbody Would Be Designated	Practical Impacts (compared to a baseline of WWH)
Warmwater Habitat (WWH)	Balanced assemblages of fish/invertebrates comparable to least impacted *regional* reference condition	Either supports biota consistent with numeric biocriteria for that ecoregion **or** exhibits the habitat potential to support recovery of the aquatic fauna	Baseline regulatory requirements consistent with the CWA "fishable" and "protection & propagation" goals; criteria consistent with U.S. EPA guidance with State/regional modifications as appropriate
Exceptional Warmwater Habitat (EWH)	Unique and/or diverse assemblages; comparable to upper quartile of *statewide* reference condition	Attainment of the EWH biocriteria demonstrated by both organism groups	More stringent criteria for D.O., temperature, ammonia, and nutrient targets; more stringent restrictions on dissolved metals translators; restrictions on nationwide dredge & fill permits; may result in more stringent wastewater treatment requirements
Coldwater Habitat (CWH)	Sustained presence of Salmonid or non-salmonid coldwater aquatic organisms; bonafide trout fishery	Bioassessment reveals coldwater species as defined by Ohio EPA (1987); put-and-take trout fishery managed by Ohio DNR	Same as above except that common metals criteria are more stringent; may result in more stringent wastewater treatment requirements
Modified Warmwater Habitat (MWH)	Warmwater assemblage dominated by species tolerant of low D.O., excessive nutrients, siltation, and/or habitat modifications	Impairment of the WWH biocriteria; existence and/or maintenance of hydrological modifications that cannot be reversed or abated to attain the WWH biocriteria; a use attainability analysis is required	Less stringent criteria for D.O., ammonia, and nutrient targets; less restrictive applications of dissolved metals translators; Nationwide permits apply without restrictions or exception; may result in less restrictive wastewater treatment requirements
Limited Resource Waters (LRW)	Highly degraded assemblages dominated exclusively by tolerant species; *should not* reflect acutely toxic conditions	Extensive physical and hydrological modifications that cannot be reversed and which preclude attainment of higher uses; a use attainability analysis is required	Chemical criteria are based on the prevention of acutely lethal conditions; may result in less restrictive wastewater treatment requirements

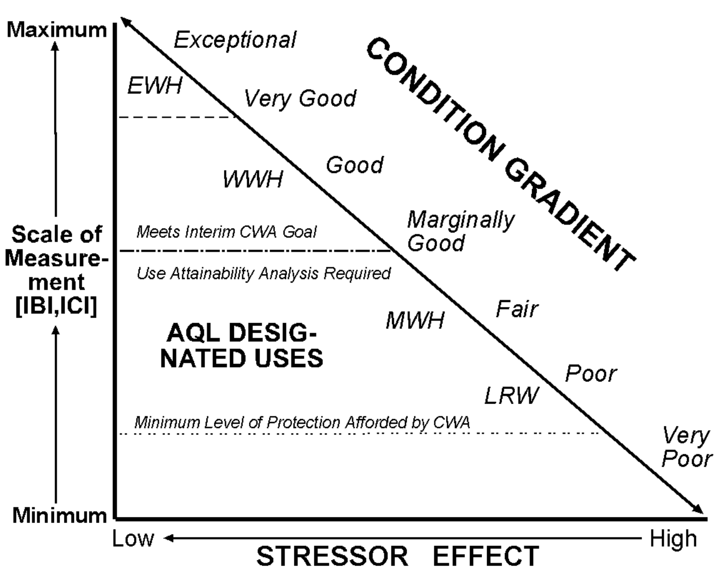


FIGURE B-3. The relationship of Ohio's tiered designated uses and numerical biological criteria to the Biological Condition Gradient.

Because of bioaccumulation concerns, many toxicant criteria are designed to protect all aquatic life uses even though they may demonstrate a graded response to the numeric biocriteria and tiered uses. For some of the heavy metals criteria where translators were developed between dissolved and total forms, concerns about the effects of potentially increased discharges of total metals resulted in a risk assessment that examined the relationships between the numeric biocriteria and total metals (Ohio EPA 1999a). This led to the derivation of "caps" on the amount of additional total metals that are permitted as a result of the dissolved metals translator process. These caps varied in accordance with the relationships demonstrated with the numeric biocriteria and tiered uses. Other parameters that do not demonstrate an empirical relationship along the BCG are not tiered. Future data exploration may well result in tiered chemical or physical criteria for stressors that are presently based on fixed, single value criteria. Such refined chemical criteria are expected to provide benefits to watershed-based management related to the prioritization of BMPs and in the application of emerging tools such as pollutant trading.

III. Establish technical program

From the outset, the implementation of tiered uses was intended to include a comprehensive and systematic monitoring and assessment program. The integration of the tiered uses with monitoring and assessment was an evolutionary development that followed the process outlined in Chapter 5 (Table 5-1; Figure 5-2) and Figure B-1.

How Does Ohio Collect Biological Data?
Ohio EPA employs a multiple chemical, physical, and biological indicators approach that utilizes each according to their most appropriate roles as indicators of stress, exposure, and response (Yoder and Rankin 1998). This approach leads to more effective regulation of pollution sources, improved assessment of diffuse and non-chemical impacts, and improves our ability to implement management strategies for successfully protecting and restoring the ecological integrity of watersheds. Key attributes that the biological indicators were developed to reflect include:

1) cost-effective collection of data
2) readily available science
3) be indicative of or extend to different trophic levels
4) integrate multiple effects and exposures
5) exhibit reasonable response and recovery times
6) be precise and reproducible
7) be responsive to a wide range of perturbations
8) be relevant to managerial and programmatic issues

Because it is impractical to monitor the entire organism assemblages present in an aquatic ecosystem, choices must be made. Ohio's choice of two organism groups (benthic macroinvertebrates and fish) is consistent with the ITFM (1992, 1995) recommendations and was done for a number of reasons. Each assemblage has been widely used in assessments and there is abundant information about their life histories, distributions, and environmental requirements. The benefit of having two different groups independently showing the same result is obvious and lends considerable strength to a bioassessment. However, differences in the responses by each group can lead to the definition of problems that might otherwise have gone undetected, underrated, or misunderstood in the absence of information from either organism group. For example, representatives of one assemblage may be able to tolerate and metabolize toxic substances that are highly detrimental to representatives of the other assemblage. The differences in recovery rates between each assemblage provide an added dimension to the understanding of how abatement processes work and document incremental changes through time. The value of such information in a risk management process should be obvious. Comparisons between the performance of fish and macroinvertebrates as arbiters of aquatic life use attainment showed non-agreement between assemblages at 33% in non-wadeable rivers, 21.2% in wadeable streams, and 28.2% in headwater streams (Yoder and Rankin 1995a). Assessments based on a single group would have overlooked proportions of the impairment that actually existed, let alone the loss of signal in diagnosing causal associations. Some of the concepts in Appendix C are based on this knowledge and experience.

Overview of the Technical Approach

The development and refinement of Ohio's biological assessment tools and criteria reflects an evolutionary process that is summarized in Figure B-1 and in Chapter 5 (Figure 5-2). The standardization of sampling and laboratory methods occurred first and illustrates the importance of the initial decisions about methods, taxonomic resolution, and professionalism early in the process (Ohio EPA 1987, DeShon 1995, Rankin 1995, Yoder and Smith 1999). From the outset of the systematic collection of biological data in Ohio, choices about sampling methods and laboratory procedures were the most important of the initial decisions that were made. These determine the attributes and characteristics of the resulting data and the usefulness and accuracy of the analytical tools and criteria that are developed. This, in turn, determines the quality of the entire approach including its ability to accurately determine biological impairments and discriminate relative quality along the BCG. Because of its primary role as a response indicator, it determines our perceptions of environmental quality and the effectiveness of our responses via water quality management programs and policies.

Sampling Methods

A number of decisions need to be made concerning the adoption of sampling methods. Decisions about sampling methods and gear, seasonal considerations, which organism groups to monitor, which parameters to measure and record, which level of taxonomy to use, etc. all were made early in the process. This was a critical juncture in the process since the decisions made here determined the effectiveness of the bioassessment effort.

The development of standardized sampling methods was the most important initial task in the implementation of Ohio's biological monitoring program. While many sampling methods and techniques existed for both macroinvertebrates and fish, many lacked adequate testing or standardization. The primary task was the testing, development, and validation of the chosen methods, which involved testing each for its ability to deliver good information at a reasonable cost. The goal was to use methods and protocols that would require 1-3 hours at a sampling site making it possible to sample several sites each day, tens of sites each week, and hundreds of sites each sampling season. A seasonal index period was also established during the summer-early fall (mid June to mid October).

For macroinvertebrates, artificial substrates were the method of choice and this was consistent with the U.S. EPA guidance of that time. The application of this method was further tested to refine the general approach in the early 1980s. A cluster of five artificial substrates bound to a concrete block are set in detectable current for a colonization period of six weeks. A dip net/hand pick sample of the surrounding natural substrates including all available habitats is collected at the time of substrate retrieval. This technique, known as qualitative sampling, employs a triangular frame dip net and can be used as a stand-alone sampling method. A site description data sheet is completed by a crew leader and includes information about the site habitat, environmental setting, and other pertinent information. Samples are retrieved, preserved in 10% formalin in the field, and transported to the laboratory for later processing. The specific methods are documented in written guidance manuals (Ohio EPA 1980, 1987, 1989b) that are codified by reference in the Ohio WQS.

Fish are collected using various wading and boat-mounted pulsed D.C. electrofishing gears, depending on the width and depth of the stream or river. These also had their origin in already available techniques, but the stratification of their use in different sizes of waterbodies was an issue that required prior testing and development. Sampling is standardized by lineal distance of stream or river and reach lengths were determined by sampling standard increments at methods test sites in the early 1980s. Fish samples are processed in the field and include identification to species, enumeration (counts and biomass) by age groups (adult, 1+, 0+), and delineation of external anomalies. A qualitative habitat assessment (QHEI; Rankin 1989, 1995) is completed over the entirety of the electrofishing reach. Fish sites are sampled once, twice, or three times within the seasonal index period, the frequency being determined by the complexity of the setting and the potential for episodic impacts. The specific methods are documented in written guidance manuals (Ohio EPA 1980, 1987, 1989b) that are codified by reference in the Ohio WQS.

Laboratory Methods
Each artificial substrate (quantitative) and natural substrate (qualitative) sample is processed in accordance with standardized procedures (Ohio EPA 1989b). This includes an initial pre-pick and visual scan for rare and large organisms, subsampling by major taxa group (mayflies, stoneflies, caddisflies, midges, others), and identification and enumeration to the lowest practicable taxonomic level. Ohio EPA staff perform both field sampling and laboratory processing.

Fish specimens that cannot be verified in the field are preserved in 10% formalin and transported to the laboratory for later processing. These are changed to 70% ethyl alcohol and identified to species. Verification of difficult specimens is performed by at least one qualified non-Ohio EPA taxonomist.

Analytical Methods
Ohio EPA analyzes biological data using routines available in the Ohio ECOS data storage, retrieval, and management system. Data is entered into Ohio ECOS following a data validation and QA/QC process to eliminate transcription and other errors. The principal indexes are based on multimetric techniques that were modified and calibrated for use in Ohio. For fish this includes the Index of Biotic Integrity (IBI; Karr 1981, Fausch et al. 1984, Karr et al. 1986) and the Index of Well-Being (IWB; Gammon 1976, Gammon et al. 1981). For macroinvertebrates it includes the Invertebrate Community Index (ICI; Ohio EPA 1987, DeShon 1995). In addition to the primary indexes, data analyses include the index metric values, relative abundance, and other aggregations of the data that exhibit ecologically meaningful patterns and information over space and time. This can include the use of multivariate analyses, parametric and non-parametric statistical techniques, and data mapping.

Staffing and Professionalism
Qualified and regionally experienced staff are employed to carry out the sampling and data analysis activities. Skilled and experienced staff direct, manage, and supervise all activities. This includes a high level of expertise in the field since many of the critical pieces of information are recorded and, to a degree, interpreted here. The same professional staff who collect the field data also interpret and apply the information derived from the data in a "cradle to grave" fashion. Thus the same staff who perform the field work also plan that work, process the data into information, interpret the results, and apply the results via assessment and reporting. Such staff, particularly those with sufficient experience, also contribute to policy and program development. The majority of data used by Ohio EPA is collected by agency staff. However, the methods and approach can be carried out by other entities and practitioners. Since 1999, Ohio EPA has operated a voluntary certification process and this will soon be mandated by the Ohio Credible Data Law.

How Does Ohio Decide What Waterbodies and Locations to Monitor?
In 1980, Ohio EPA initiated an intensive watershed survey design that included chemical/physical and biological assessments or surveys. A biological and water quality survey, or "biosurvey," is an interdisciplinary monitoring effort coordinated on a waterbody specific or watershed scale. The effort may involve a relatively simple setting focusing on one or two small streams, one or two principal stressors, and a handful of sampling sites or a much more complex effort including entire drainage basins, multiple and overlapping stressors, and tens of sites. Through the 1980s, Ohio EPA conducted biosurveys in 6-10 different study areas with an aggregate total of 250-300 sampling sites sampled/year.

- Rotating basin approach for determining annual monitoring activities.
- Correlated with NPDES permit schedule.
- Supports annual WQS use designation rule-making.
- Aligned with 15 year TMDL schedule in 1998.

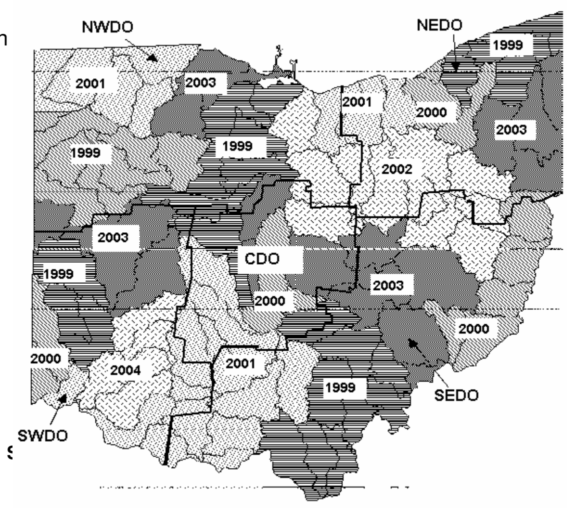


FIGURE B-4. Five-year basin approach for determining annual watershed monitoring and assessment activities and correspondence to support major water quality management programs.

While the purpose of these surveys was to support multiple program objectives, the schedule of water quality management program outputs was not always coordinated with the biosurvey schedule. In 1990, this process was formally coordinated beginning with a revision to the schedule for reissuance of major and significant NPDES permits. Ohio EPA formally adopted a five year basin approach in which biosurveys were scheduled two years in advance of the reissuance of NPDES permits (Figure B-4). The rotating basin approach proved its utility in two other instances. The first was in support of Ohio nonpoint source assessment in 1990, and the second when TMDLs became a major priority in 1998. The latter was seamlessly integrated into the rotating basin approach (Ohio EPA 1999b). In the 1990s, the demand for the watershed assessments increased further with up to 700 sites being sampled within 10-12 study areas in some years. The process of program integration was further institutionalized with a structured process for selecting watersheds, planning the monitoring, and analyzing and reporting the results (Table B-5).

TABLE B-5. Important timelines and milestones in the planning and execution of the rotating basin approach conducted annually and since 1990 by Ohio EPA.

Milestone	Timeline
December - February: (Months 1-3)	Initial screening of the major hydrologic areas takes place by soliciting input from the various program offices and other stakeholders.
February - March: (Months 3 thru 4)	Final prioritization of issues and definition of specific study areas. Resource allocation takes place and study team assignments are made.
March - May: (Months 4 thru 5)	Study planning takes place and consists of detailed map reconnaissance, review of historical monitoring efforts, and initial sampling site selection by the study team. Final study plans are reviewed and approved.
May - June: (Months 5 thru 6)	Final study plans are used to develop logistics for each field crew. Preparations are made for full-scale field sampling.
June - October: (Months 6 thru 10)	Field sampling takes place with field crews operating somewhat independently on a day-to-day basis, but coordinated by the study plan and the team leader. Study team communication takes place as necessary, especially to resolve unexpected situations.
October - February: (Months 10 thru 14)	Laboratory sample analysis takes place for chemical and biological parameters. Raw data is entered into databases for reduction and analysis. The study team meets to review the information base generated by the field sampling and to coordinate the data analysis and reporting effort.
November - May: (Months 11 thru 17)	Information about indicator levels 3-6 is retrieved, compiled, and used to produce analyses that will support the evaluation of status and trends and causal associations within the study area. Integration of the information (*i.e.,* assessment) is initiated.
May - December: (Months 17 thru 24)	The assessment process is completed by producing working copies of the assessment for review by the study team and a final edit for an internal peer review. Final assessment approved by management for use within and outside of Ohio EPA. It is used to support 305b /303d, NPDES permitting, water quality standards (*e.g.,* use designation revisions), and other programs where surface water quality is of concern.

Each biosurvey is designed and conducted to meet three major objectives:

1) determine the extent to which use designations assigned in the Ohio WQS are either attained or not attained;
2) determine if use designations assigned to a given waterbody are appropriate and attainable; and
3) determine if any changes in key ambient biological, chemical, or physical indicators have taken place over time, particularly before and after the implementation of point source pollution controls or best management practices.

The data gathered by a biosurvey is processed, evaluated, and synthesized in a biological and water quality report. Each biological and water quality study contains a summary of major findings and recommendations for revisions to WQS (e.g., Table B-6), future monitoring needs, or other actions which may be needed to resolve existing impairment(s) of designated uses. At the same time, the systematic execution of basin surveys builds a long-term database over space and time, creating and sustaining a resource of the development and improvement of tools, criteria, policies, and legislation (Figure B-5).

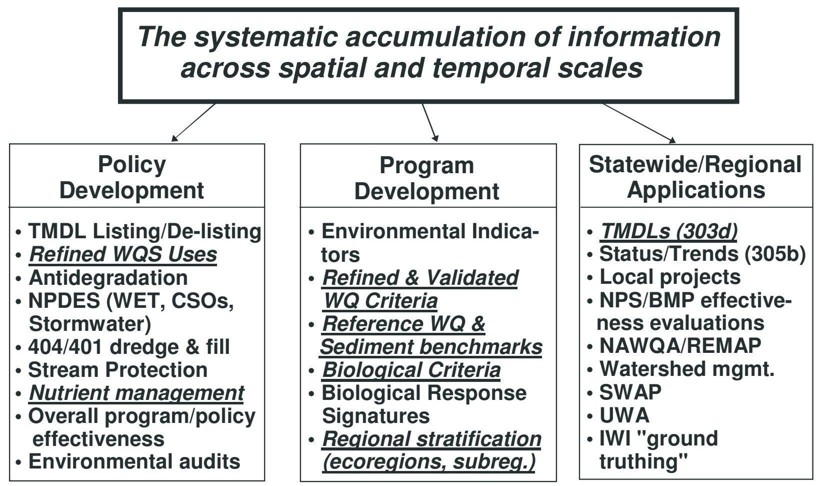


FIGURE B-5. Strategic support provided over time by systematic monitoring and assessment; functions related to the implementation of TALUs are italicized and underlined.

The recommendations for use designation revisions are a direct result of the biological and water quality assessment. Uses are designated on demonstrated potential to attain a particular use based on the following sequence (in order of importance):

1) attainment of the biocriteria (if attaining WWH or higher – attainment of EWH is required to be designated as EWH); and

2) if a WWH biocriterion is not met, the habitat potential determined by the Qualitative Habitat Evaluation Index (QHEI; Rankin 1989, 1995) and an associated assessment of warmwater: modified habitat attributes is used to determine the potential to attain WWH.

For uses less than WWH (i.e., MWH or LRW), a use attainability analysis is required and includes consideration of the factors that essentially preclude WWH attainment including the feasibility of restoring the waterbody. A use attainability analysis requires the following information:

1) the present attainment status of the waterbody based on a biological assessment performed in accordance with the requirements of the biocriteria, the Ohio WQS, and the Five-Year Monitoring Strategy (the latter pertains to adequacy of spatial design);

2) a habitat assessment to evaluate the potential to attain at least WWH; and

3) a reasonable relationship between the impaired status and the precluding human-induced activities based on an assessment of multiple indicators used in their appropriate indicator roles and a demonstration consistent with 40CFR Part 131.10 [g][1-6].

In the example from the Big Darby Creek watershed assessment conducted in 2000, all of the streams and segments listed in Table B-6 were sampled in accordance with Ohio EPA's geometric and intensive survey design. A number of the streams in Table B-6 were originally assigned aquatic life use designations in the 1978 and 1985 WQS based largely on best professional judgment or by tributary membership, while others were not yet designated. The current biological assessment methods and numerical biocriteria did not exist at that time. Most of the larger tributaries and the mainstem were previously designated based on biosurveys of specific segments and streams in 1979, 1981, 1988, and 1992. The use designations of most of the mainstem and some of the major tributaries were resolved by those efforts. However, many of the smaller streams in this watershed were evaluated for the first time using a standardized biological approach in 2000. Ultimately, the designations for each stream and river segment are based on direct sampling and assessments of each individual waterbody and the processes previously described. Extrapolation of sampling results for this and other purposes (e.g., status assessment) is minimal and occurs only within individual waterbodies. The application of the geometric watershed and intensive survey design included all tributaries and resulted in the addition of 26 previously unlisted and/or undesignated streams. Of these 26 streams, four were designated EWH, 18 as WWH, four as MWH, and two as LRW; an additional five stream segments were simultaneously designated CWH. Under the 1978 WQS, all 26 tributaries would have been designated as EWH by virtue of their tributary membership in the Big Darby watershed. This was extended to only the 19 named tributaries in the 1985 WQS, of which nine were later changed based on earlier biosurvey data. This example illustrates the comparative lack of accuracy in extrapolating uses by tributary membership within a watershed and the need to sample and assess individual streams for use designation purposes.

TABLE B-6. Summary of recommendations for use designations in the Big Darby Creek watershed based on a biological and water quality assessment completed in 2000. Symbols are listed for the existing designation/recommended designation (_ - undesignated; + - verified by biosurvey; * - unverified default designation from 1978 or 1985 WQS).

	Use Designations												
		Aquatic Life Habitat						Water Supply			Recreation		
Water Body Segment	SRW	WWH	EWH	MWH	SSH	CWH	LRW	PWS	AWS	IWS	BW	PCR	SCR
Big Darby Creek (02-200) [a] - Headwaters to RM 79.2		*	+			+			*+	*+		*+	
- RM 79.2 to mouth			*+						+	+		+	
Flat Branch (02-223) (RM 78.48)[b]			*	+					+	+		+	
Tributary to Flat Branch (02–365) (RM 1.5)				_+					_+	_+		_+	
Little Darby Creek (02-251) (RM 78.34) RM 3.5 to mouth			+			+			_+	_+		_+	
U.T. to B. Darby Cr. (02-361) (RM 74.91) RM 0.75 to mouth			_+						_+	_+			_+
Spain Creek (02-222) (RM 74.3) - Headwaters to RM 5.0		+	*			+			+	+		*+	
RM 5.0 to mouth			*+			+			+	+		*+	
Pleasant Run (02-221) (RM 72.01)			*+						+	+		*+	

Water Body Segment	Use Designations												
		Aquatic Life Habitat						Water Supply			Recreation		
	SRW	WWH	EWH	MWH	SSH	CWH	LRW	PWS	AWS	IWS	BW	PCR	SCR
U.T. to Big Darby Creek (02-360) (RM 69.4) RM 1.8 to mouth		_+							_+	_+			_+
Hay Run (02-220) (RM 67.6) RM 1.1 to mouth			_+						+	+		*+	
Prairie Run (02-219) (RM 63.84)							_+		_+				_+
Buck Run (02-209) (RM 63.74)		+	*						+	+		_+	
Robinson Run (02-207) (RM 53.69)		+	*						+	+		*+	
Sweeney Run (02-357) (RM 52.11) RM 1.7 to mouth		_+							_+	_+		_+	
Sugar Run (02-206) (RM 50.92) - Headwaters to RM 7.0			*	+					+	+		*+	
- RM 7.0 to mouth		+	*						+	+		*+	
U.T. to Sugar Run (02-358) (RM 7.39)				_+					_+	_+		_+	
Worthington Ditch (02-2356) (RM 50.62) RM 0.4 to mouth		_+							_+	_+		_+	
Ballenger-Jones Ditch (02-355) (RM 49.68) RM 3.72 to mouth		_+							_+	_+		_+	
Yutzy Ditch (02-364) (RM 47.1) RM 1.38 to the mouth		_+							_+	_+		_+	
Fitzgerald Ditch (02-272) (RM 44.96) RM 1.75 to mouth		+							_+	_+		_+	
Little Darby Cr.(02-210) (RM 34.1) Headwaters to RM 36.9			*+			+			+	+		+	
Little Darby Cr.(02-210) (RM 34.1) RM 36.9 to mouth			*+						+	+		+	
Clover Run (02-218) (RM 39.8)		_+							*+	*+			_+
Lake Run (02-216) (RM 36.9)		_+							*+	*+		*+	
Jumping Run (02-217) (RM 3.9)		_+							*+	*+		*+	
Treacle Creek (02-213) (RM 31.3)			*+						*+	*+		*+	
Howard Run (02-215) (RM 5.4)			*+						*+	*+		*+	
Proctor Run (02-214) (RM 3.69)			*+						*+	*+		*+	
Barron Creek (02-212) (RM 24.4)		_+							*+	*+			_+
Wamp Ditch (02-363) (RM 23.0)		_+							_+	_+			_+
Spring Fork (02-211) (RM 17.46)			*+						*+	*+		*+	
Bales Ditch (02-362)(RM 3.64) RM 1.72 to mouth		_+							_+	_+			_+
Smith Ditch (02-353) (RM 31.69)			_+						_+	_+		_+	
Tributary to Smith Ditch (02-354)(RM0.06)			_+						_+	_+		_+	
Gay Run (02-298) (RM 26.48)		_+							_+	_+		_+	
Hellbranch Run (02-204) (RM 26.1) Headwaters to RM 5.0		+	*						+	+		+	
Hellbranch Run (02-204) (RM 26.1) RM 5.0 to mouth			*+						+	+		*+	
Hamilton Ditch (02-259) (RM 11.19) -Hdwtrs to Feder Rd.				_+					*+	*+		_+	

Water Body Segment	Use Designations												
		Aquatic Life Habitat						Water Supply			Recreation		
	SRW	WWH	EWH	MWH	SSH	CWH	LRW	PWS	AWS	IWS	BW	PCR	SCR
Feder Rd. to mouth		_+							*+	*+		_+	
Clover Groff Ditch (02-245) (RM 11.19 - Hdwtrs to Feder Rd.				_+					*+	*+		_+	
Feder Rd. to mouth		_+							*+	*+		_+	
Springwater Run (02-203) (RM 24.0)		+	*						+	+		*+	
U.T. to Big Darby Creek (02-352) (RM 23.77)		_+							_+	_+			_+
U.T. to Big Darby Creek (02-270) (RM 20.2)		_+							_+	_+		_+	
U.T. to Big Darby Creek (02-366) (RM 18.41)		_+							_+	_+			_+
Greenbrier Creek (02-202) (RM 16.75)		+	*						+	+		*+	
Georges Creek (02-201) (RM 14.4)		+	*						+	+		*	_+
Lizard Run (02-273) (RM 12.93)							_+		+	+			_+

a - River code of the river or stream segment; b - River Mile of the confluence point with applicable receiving stream

While the principal focus of a biosurvey is on the status of aquatic life uses, the status of other uses such as recreation and water supply, as well as human health concerns are also addressed (Table B-6). The findings and conclusions of a biological and water quality study may factor into regulatory actions taken by Ohio EPA (e.g., NPDES permits, Director's Orders, the Ohio Water Quality Standards [OAC 3745-1]), and are eventually incorporated into Water Quality Permit Support Documents (WQPSDs), State Water Quality Management Plans, the Ohio Nonpoint Source Assessment, and the Integrated Report (combined 303[d] and 305[b] report). Periodic rulemakings are conducted to incorporate the use revision recommendations into the Ohio WQS, thus resolving the issue *prior to* the application of water quality management (see Figure 5-1, U.S. EPA's Water Quality Management Cycle). Figure B-6 summarizes the number of stream and river segments (mostly whole streams) where aquatic life uses have been revised as the result of a biological and water quality assessment in Ohio since 1978. This became a routine practice once the assessment criteria and decision-making process

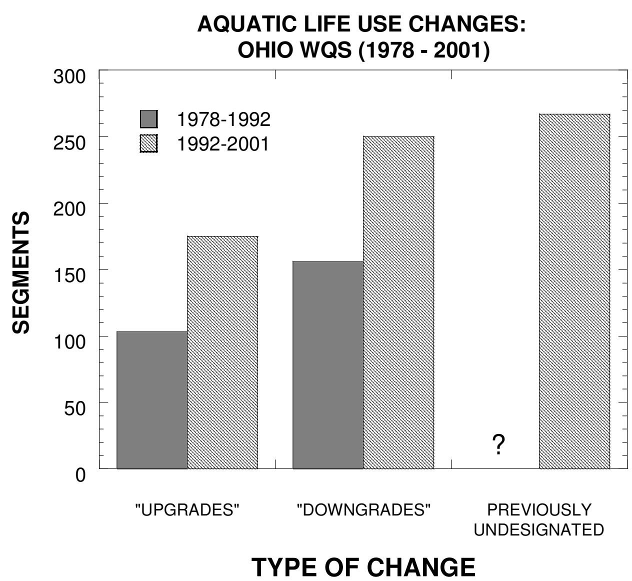


FIGURE B-6. The number of individual stream and river segments in which aquatic life use designations were revised during 1978-1992 and 1992-2001. Cases where the use was revised to a higher use are termed "upgrades" and cases where a lower use was assigned are termed "downgrades". Previously undesignated refers to streams that were not listed in the 1985 WQS, but which have been added via the Five-Year Basin Approach to monitoring and assessment.

for UAAs were established earlier in the assessment process. It required the development of reliable tools, particularly for determining status, assessing habitat, and determining causal associations, all of which are part of the developmental process described in Figure 5-2. The terms “upgrade” and “downgrade” are used figuratively here and in Figure B-6 as descriptors of the direction of change from the default use to that produced by a standardized assessment process. The majority of these changes are from the baseline of the original designations made in 1978 or 1985 without the benefit of systematic monitoring and assessment data, numerical biocriteria, and refinements in the process that occurred in the late 1980s. Thus, the original use designations are merely being “corrected” to the appropriate use based on a standardized process and more robust criteria and assessments.

Monitoring and assessment information, when based on a sufficiently comprehensive and rigorous system of environmental indicators, is integral to protecting human health, preserving and restoring ecosystem integrity, and sustaining a viable economy (ITFM 1992). Such a strategy is intended to achieve a better return on public and private investments in environmental protection and natural resources management. More and better monitoring and assessment information is needed to answer the fundamental questions about the condition of our water resources and to shape the strategies needed to address both existing and emerging problems within the context of watershed-based management. These principles have guided the development of surface water monitoring and assessment at Ohio EPA for the past 25 years and will continue to do so in the future.

IV. Develop and validate quantitative thresholds

The lack of adequate and reliable decision criteria for biological assessment has historically limited its usefulness, reliability, and wider acceptance in water quality management. In 1980, Ohio EPA developed an initial set of decision criteria for fish and macroinvertebrate assemblages that consisted of narrative quality ratings based in part on numerical biological index “guidelines” (Tables B-1 and B-2). These were intended to more directly reflect and assess the ecological goals espoused by the tiered aquatic life uses adopted in 1978. These early narrative biocriteria were comprised of contemporary measures such as taxa richness, indicator guilds, the Shannon diversity index, and the Index of Well-Being (Gammon 1976). Attainable expectations for a set of narrative community attributes were based on Ohio’s experience with sampling approximately 150-200 sites statewide. This approach was used between 1980 and 1987 and was applied uniformly on a statewide basis. As the technology did not yet exist, no effort was made to account for background variability by using landscape partitioning frameworks such as ecoregions.

The narrative classification system consisted of assigning narrative quality ratings such as exceptional (consistent with the Exceptional Warmwater Habitat use), good (Warmwater Habitat use), fair, and poor. Exceptional and good met the goals of the Clean Water Act while fair and poor reflected a failure to attain those goals (Tables B-1 and B-2). The purpose of this narrative classification system was essentially two fold: 1) to provide an objective, systematic basis for assigning aquatic life uses to surface waters; and 2) to provide an objective, standardized approach for determining the magnitude and severity of aquatic life impairments for assessment purposes. Considerable judgment was used in applying these early narrative biological criteria on a site-specific basis and the system was characteristic of between a level 2 and 3 program *(See Appendix C)*. The aggregate impact of these assessments played a major role in setting and evaluating WQS use designations, designing water quality management plans, and developing advanced treatment justifications for municipal sewage treatment plants. These criteria also provided a basis for designating stream and river segments as attaining, partially attaining, or not attaining designated aquatic life uses in the 1982, 1984, and 1986 Ohio EPA 305b reports. They were, however, inherently prone to under-estimating impairment (DeShon 1995).

Regionally Referenced Numerical Biological Criteria

In 1986, a major effort was undertaken to develop regionally referenced and calibrated numeric biological criteria using a statewide set of regional reference sites. This was spurred by the Ohio Stream Regionalization Project in which the application of Omernik's (1987) ecoregions and the regional reference site concept (Hughes et al. 1986) was tested. For the fish assemblage, the Index of Well-Being was modified (Ohio EPA 1987) and the Index of Biotic Integrity (IBI; Karr 1981, Karr et al. 1986) was added. For macroinvertebrates, the Invertebrate Community Index (ICI; Ohio EPA 1987, DeShon 1995) replaced the narrative evaluations used previously. The IBI and ICI consist of metrics that include community production, function, tolerance, and reproduction in an aggregated index. This provides for a more rigorous, ecologically oriented approach to assessing aquatic community health and well-being. The process of deriving the numerical biological criteria is described more extensively in Ohio EPA (1987, 1989a,b) and Yoder and Rankin (1995a).

The derivation of the current numerical biological criteria is based on the biological "performance" that is demonstrated at least impacted, regional reference sites. This is consistent with the operational definition of biological integrity as defined by Karr and Dudley (1981), which provides the theoretical basis for this framework. The numerical biological criteria resulting from the application of this framework represent the assemblage performance that can reasonably be attained given contemporary background conditions. Although these do not emanate from an attempt to define "pristine," pre-Columbian conditions, the design framework includes a provision to "maintain" the biocriteria by continually resampling the reference sites – reference condition is monitored so that all reference sites are resampled once each decade. This promotes the periodic and orderly reassessment of reference condition and the database that drives the calibration of the biological indexes and the derivation of the numeric biocriteria. Furthermore, the knowledge base used in the development of the multimetric indexes includes an awareness of pre-settlement faunas and their characteristics. This is entirely consistent with the BCG and the description of attributes from "as naturally occurs" to an increasingly disturbed state. Thus, if pristine conditions do return this would be reflected by the periodic adjustments to the multimetric indexes, their calibration, and/or the numerical biological criteria.

Biological criteria in Ohio are based on two principal organism groups, fish and macroinvertebrates. Numerical biological criteria for rivers and streams were derived by utilizing the results of sampling conducted at more than 400 reference sites that represent the "least impacted" conditions within each ecoregion (Ohio EPA 1987, 1989a). This information was then merged within the existing framework of tiered aquatic life uses to establish attainable, baseline biological assemblage performance expectations on a regional basis. Biological criteria vary by ecoregion, aquatic life use designation, site type, and biological index (Figure B-2).

The framework within which biological criteria were established and used to evaluate Ohio rivers and streams includes the following major steps:

- selection of indicator organism groups;
- establish standardized field sampling, laboratory, and analytical methods;
- selection and sampling of least impacted reference sites;
- calibration of multimetric indexes (e.g., IBI, ICI);
- set numeric biocriteria based on attributes specified by each tiered aquatic life use designation;
- reference site re-sampling (10% of sites sampled each year beginning in 1990); and,
- making periodic (i.e., once per 10 years) adjustments to the multi-metric indexes, numeric biocriteria, or both as determined by reference site resampling results (Note: this latter step has yet to be undertaken by Ohio EPA).

The major steps in the biological criteria calibration, derivation, and application process are summarized in Figure B-7. The process integrates the technical process of index derivation and calibration with narrative statements about the desired biological assemblage condition and regionalization (e.g., ecoregions). This latter step is particularly important as it is needed to stratify regional landscape variability within a tractable framework. Figure B-7 portrays the calibration of the IBI for wading sites. A similar stepwise procedure was used to calibrate the Invertebrate Community Index for macroinvertebrates (Ohio EPA 1987, DeShon 1995) and the IBIs for the headwater and boatable site types. Once reference sites are selected and sampled (Step 1 in Figure B-7) the biological data is first used to calibrate the IBI (Step 2) and ICI. For fish three different IBIs were derived, one each for headwaters, wading (Step 3), and boat sites. The reference site IBIs are then used to establish numerical biological criteria (Steps 4 and 5). A notched box-and-whisker plot method was used to analyze the distribution of IBIs by ecoregion (Step 4). These plots contain sample size, medians, ranges with outliers, and 25th and 75th percentiles. Box plots have one important advantage over the use of means and standard deviations (or standard errors) because they do not assume a particular distribution of the data. Furthermore, outliers (i.e., data points that are two interquartile ranges beyond the 25th or 75th percentiles) do not exert an undue influence as they can on means and standard errors. In establishing biological criteria for a particular area or ecoregion we attempted to represent the "typical" biological community performance, not the extremes and outliers. These can be dealt with on a case-by-case or site-specific basis, if necessary. Once numerical biological criteria are determined, they are then used in making assessments of specific rivers and streams (Step 6).

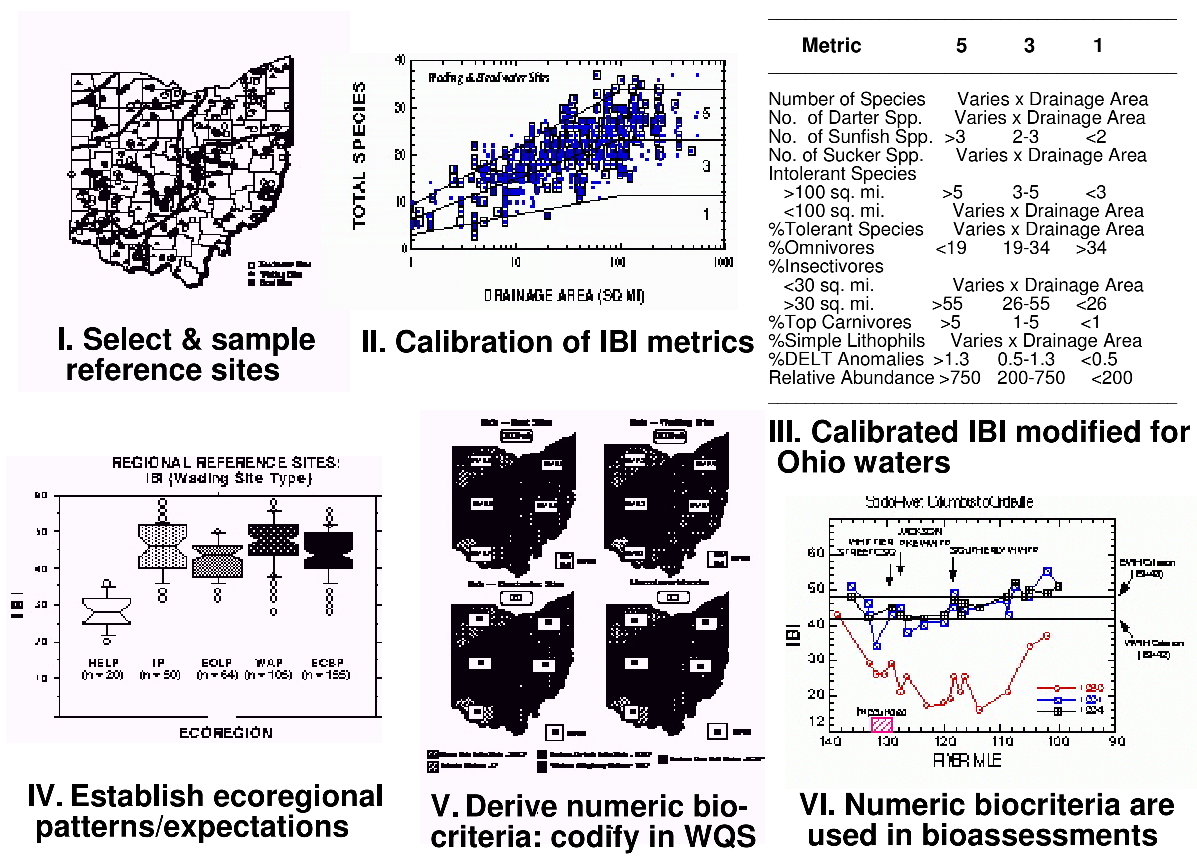


FIGURE B-7. The major steps of the Ohio EPA numeric biological criteria calibration and derivation process leading to their application in biological and water quality assessments; this example is for the Index of Biotic Integrity (IBI) for wading sites.

The outcome is a systematic process for measuring the essential products of aquatic structure and function that represent symptoms of ecosystem health. BCG derived and calibrated numeric biocriteria provide tangible measures of aquatic assemblages by which ecosystem health and well-being can be inferred. The tangible products of healthy watersheds are desirable biomass, water quality that is suitable for all uses, and an ability to assimilate background inputs that do not alter the key characteristics or processes associated with the aquatic assemblages detailed in the BCG (Table B-7). The key indicators of each are biological assemblage performance consistent with the designated use (measured by the biological indexes and compared to the numeric biocriteria) and chemical and physical quality comparable to least impacted regional reference conditions and other acceptable exposure thresholds.

TABLE B-7. The tangible products that are symptomatic of aquatic ecosystem health and the measurable biological, chemical, and physical indicators of healthy and degraded aquatic systems.

Tangible "Products"	Healthy	Degraded
Biomass	Desirable forms (quality biodiversity, game fish, birds, mammals, inverts., plants, algae, microbes)	Undesirable forms (low quality biodiversity, nuisance abundances, tolerant species dominate)
Water Quality	Comparable to regional reference	Poorer than regional reference
Assimilative Capacity	Processes background runoff and materials without adverse changes in biota	Inability to process background inputs due to reduced capacity to biologically and physically process excess materials
Measurable Indicators:		
Biological assemblages	Meet or exceed numeric biocriteria for TALU	Does not meet biocriteria for TALU; response varies by impact type and severity of impairment
Chemical indicators	Meets numeric criteria (some are TALU based) and is within reference thresholds	Exceeds numeric criteria and/or reference thresholds
Physical Indicators	Provides essential habitat attributes and hydrology	Degraded habitat and altered hydrology

What Process Was Used to Adopt Biocriteria in the Ohio WQS?

The adoption of numeric biocriteria and tiered uses in the Ohio WQS has been an evolutionary process over the preceding 25 years. There were many important events that determined the make-up and acceptance of the biocriteria and TALU in Ohio. These milestones are summarized in Table B-8. Some of the key events that resulted in a wider acceptance of the present day biocriteria and tiered uses were the legal proceedings on the use changes that occurred in the lower Cuyahoga River in 1988 and Ottawa River in 1989. Ohio EPA adopted a recommendation that the use designation of the Cuyahoga River mainstem be changed from a Limited Warmwater Habitat use designation to Warmwater Habitat based on biological and water quality surveys conducted between 1984 and 1987 and the ensuing UAA process. The former use was adopted in 1978 as a variance for specific point source derived pollutants. The biological assessments concluded that while the mainstem was severely impaired, the potential to attain WWH with achievable water quality based management of point sources was supported by the habitat assessment that showed a sufficiently intact habitat. This was eventually resolved via a legal process that included appeals of the initial decision up to the Ohio Supreme Court.

TABLE B-8. Key events and milestones that occurred in the evolutionary development, adoption, and implementation of biological assessments, numeric biocriteria, and tiered aquatic life uses in Ohio between 1974 and the present.

YEAR	MILESTONE	DESCRIPTION
1974	First Ohio WQS	General use, few numeric criteria, narrative "free froms"
1978	Initial TALUs	Tiered uses adopted, specific chemical criteria
1980	Narrative "biocriteria"	First organized approach to biological assessment; systematic monitoring & assessment
1983-4	Stream Regionalization Project	Testing and validation of Omernik's ecoregions and reference site concepts in Ohio
1986-7	Derivation of numeric biocriteria	Statewide data collected to date was used to develop, derive, and calibrate numeric biocriteria based on multimetric indexes; biocriteria "User Manuals" published
1987	Biocriteria proposed in WQS	Initial proposal for numeric biocriteria
1987-89	Hearings on Cuyahoga River use change	Litigation of revision of a segment of the river form LWH to WWH; regulated entities Contested basis for the "upgrade"; the first test of the technical and policy aspects of the numeric biocriteria and TALU implementation; resolved at Ohio Supreme Court
1989	Hearings on Ottawa River use change	Litigation of revision of a segment of the river from LWH to WWH; regulated entities challenged; issue settled after Cuyahoga case ruling; led to more stringent regulation of point and nonpoint sources.
1990	Biocriteria adopted	Numeric biocriteria and refinements to TALUs were formally adopted in WQS
1990	Five-Year Basin Approach	A rotating basin approach that integrated key WQ management program outputs (e.g., NPDES permits) was initiated; use changes processed in annual rulemakings
1991	Internal training and orientation	All water program staff receives training in WQS, monitoring & assessment, modeling, and permit development and their integration.
1995	Lake Erie Bioassessment	Biological assessment methods and indexes developed for application to Lake Erie near shore and lacustuary habitats
1998	Wetlands bioassessment methods and biocriteria	Bioassessment methods and narrative criteria were developed for wetlands; includes various standardized assessment methods (beyond delineation) and a classification scheme.
1998	TMDL development process & schedule	TMDL development was integrated into the Five-Year Basin Approach ad schedule through 2015
1999	Re-sampling of regional reference sites	First re-sampling of regional reference sites was completed via the Five-Year Basin Approach
2003	Primary Headwater Habitat	Assessment and classification scheme for primary headwater streams that are not included in the existing numeric biocriteria are developed as a result of stream management applications.
2003	Ohio River	ORSANCO develops biological assessment tools and indexes as a precursor to numeric biocriteria for the Ohio R. mainstem.

Data collected via follow-up monitoring between 1984 and 2000 shows that attainment of the WWH biocriteria is increasing in the mainstem and proving the validity of both the WWH designation and the water quality based pollution abatement that the redesignation spurred (Figure B-8). A similar case involving the Ottawa River was resolved when this legal decision was made. No other appeals of the hundreds of use changes that have been made since that time have been filed. The systematic process of resolving use designation issues ahead of water quality management actions (permitting, listing, funding, planning) has proceeded as one of the most important outcomes of the Five-Year Basin Approach since that time. The next major milestone for the program will be the analysis of the first set of reference sites re-sampling that took place in the 1990s. In addition, level IV subregions have been delineated, which offers an additional level of potential stratification to the biocriteria derivation process.

The developments that occurred in the late 1990s including biological assessment and classification schemes for wetlands, Lake Erie near shore and lacustuary habitats, primary headwater stream habitat, and the Ohio River all happened as a result of the ground work laid in the 1980s for streams and rivers. It illustrates the natural growth process that can occur once the fundamentals of the approach are developed, tested, and adopted.

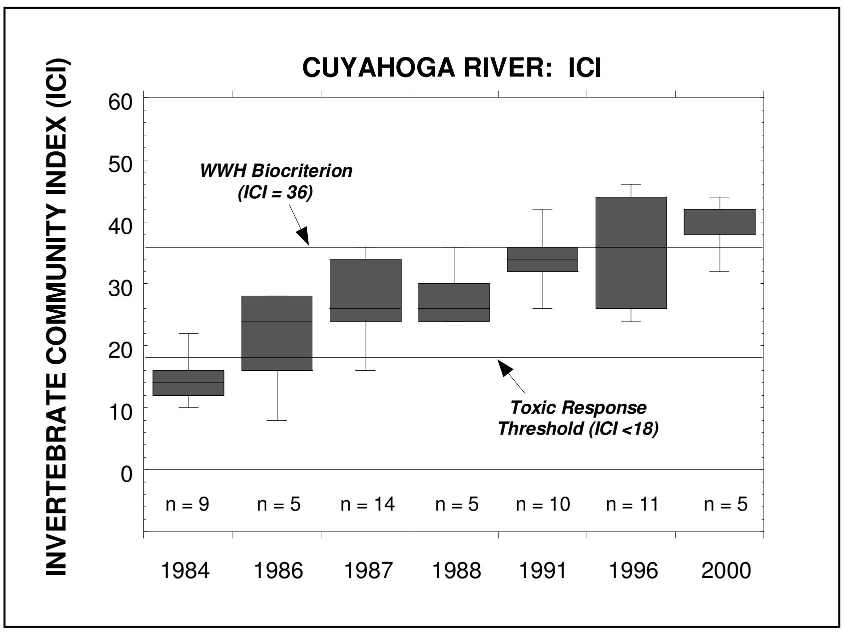


FIGURE B-8. Box-and-whisker plots of Invertebrate Community Index (ICI) results in the mainstem of the Cuyahoga River between Akron and Cleveland between 1984 and 2000.

Appendix C

Technical Guidelines: Technical Elements of a Bioassessment Program

(SUMMARY OF DRAFT DOCUMENT)

[This document has undergone review by State and U.S. EPA Regional biologists and managers. Data analyses are currently being conducted to refine certain technical elements (e.g., subsampling level, taxonomic resolution, spatial array of sites) that determine the level of rigor. A revised version will be prepared for a more comprehensive review by States and Tribes prior to finalization. A draft document will be available to the public in 2006.]

What are these technical guidelines and what is the purpose of the document?

This document is intended primarily for use by State and Tribal program managers and staff responsible for monitoring and assessment and WQS programs. States and Tribes can use this information to assess and communicate the precision of biological programs and, if deemed necessary, to refine and modify those programs. As States and Tribes increasingly use biological assessments and criteria to refine designated aquatic life uses, the need to recognize and communicate the level of precision of the biological program takes on greater importance. In addition, when the majority of States are in various stages of developing and improving their biological assessment programs, States and Tribes can use the type of detailed guidelines and milestones provided in this document to evaluate their progress.

Bioassessment is a major component of monitoring and assessment programs that include other chemical, physical, and environmental measures and indicators (ITFM 1992, 1995; Yoder and Rankin 1998). This document describes the critical, or key, technical attributes and processes of State and Tribal biological assessment programs. State and Tribal monitoring programs can also use the technical information presented in this document as a procedural template for evaluating the technical elements of their chemical and physical monitoring and assessment approaches. Ultimately, the integration of chemical and physical assessment with biological assessment will provide information to help States and Tribes better determine priorities and make more informed management decisions. State and Tribal programs can achieve appropriate levels of precision in their monitoring and assessment programs using currently available methods and technologies, and these approaches will produce a sufficiently accurate, comprehensive, and cost-effective program capable of supporting all water quality management programs.

What are the key technical elements of a bioassessment program?

There are 12 key technical elements that compose three basic methodological components: sampling design, methods, and data interpretation (see box at right). To better understand each technical element in a bioassessment program, it is important to

The 12 Key Technical Elements of a Bioassessment Program

Sampling Design Component

1. *Temporal Periodicity* of the sampling
2. *Spatial Coverage* of the sites within the area of interest
3. *Natural Classification* of the waterbodies as a framework for assessment
4. *Regional Reference Condition* development
5. *Reference Sites Selection Criteria*

Methods Component

6. Number and kinds of *Indicator Assemblages*
7. Methods for *Sample Collection*
8. Methods for *Sample Processing*

Data Interpretation Component

9. Attention to *Ecological Attributes* for indicators
10. Calibration of *Biological Endpoints*
11. *Diagnostic Capability* of the indicators
12. Use of *Professional Review* of documentation and methods

articulate the underlying rationale for each. The technical elements will be described in detail in the draft document that is being prepared for review.

How do environmental managers use these guidelines to evaluate the precision of their bioassessment program?

Included in the *Technical Guidelines* document is a checklist that enables managers and technical staff to evaluate their program's level of rigor for each of the 12 key technical elements. The checklist includes four levels of rigor, with Level 4 being the most rigorous. For an overall assessment of a water quality agency's bioassessment program, a checklist should be completed for each assemblage and waterbody ecotype, as bioassessment programs may have different levels of rigor for different waterbody ecotypes. It is important for the water quality agency to determine and reconcile these for management purposes since differing levels of rigor provide different levels of confidence in decision-making.

Evaluation of a program's level of rigor should be conducted collaboratively with State and Tribal technical staff and managers. Documentation will support completion of the checklist regarding aspects of the technical elements. Some variation between different elements will likely occur in terms of performance level (i.e., one element may receive a Level 4, while another is determined to be Level 2). Therefore, a scale that combines the rating of all elements will provide an overall indication of bioassessment program rigor. This cumulative evaluation provides a detailed analysis of the strengths and weaknesses of the comprehensive bioassessment and biocriteria program. In this rating system, we have considered all elements to be of equal weight. However, the data acquisition (sampling, processing) and treatment (analysis) phase is the linchpin of any program. One of the questions under discussion in preparation of this draft document is how to evaluate the influence of these particularly key elements.

What are the implications of having a bioassessment program with a high level of rigor?

The rigor and quality of biological assessments may vary among water resource agencies. The quality of the biological data is integral to effectively and accurately answering questions about condition, protection, restoration, or other management decisions regarding surface water resources. For example, bioassessment data obtained using a low level of rigor may provide a lesser degree of resolution needed to differentiate many stressor effects from natural variability.

The guidelines focus on four levels of rigor, where Level 4 is the most rigorous and provides the highest quality of data. The lower levels of rigor may detect and describe severely altered waters, and to a more limited extent, waterbodies in the best condition. As the level of rigor increases, the ability to discern more precisely different levels of biological condition increases. Figure C-1 illustrates the theoretical performance of the four levels of rigor of bioassessment techniques in assessing condition and the level of confidence in those assessments.

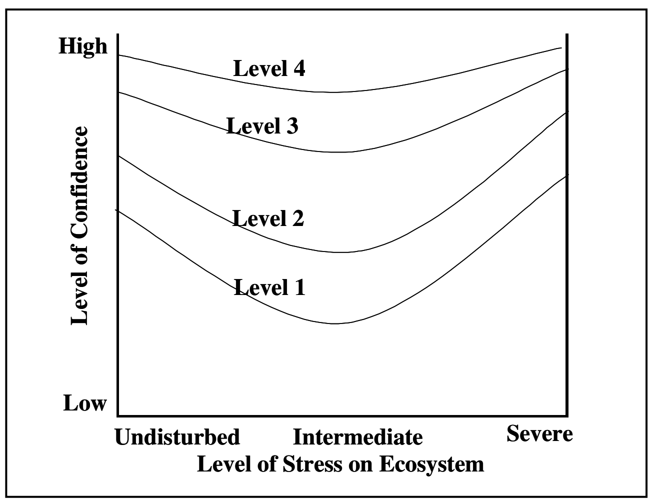


FIGURE C-1. Conceptual illustration of confidence in detecting different stress levels as a function of assessment rigor (Levels 1-4 with 4 being most rigorous).

Detecting and quantifying intermediately stressed sites, accurately describing associated causes and sources, and measuring along a stressor gradient will be done more accurately and with more confidence as the level of rigor increases. In Figure C-1, Level 4 provides the highest confidence in the biological assessment along the stressor gradient. Progressively less rigorous methods provide higher confidence in the assessment at the extremes of the stressor continuum, often only useful for status assessments. The difference in levels of rigor will be more apparent in applications requiring diagnostic capability. By first identifying the level of rigor attained by each of the key elements and the overall approach, States and Tribes can better use the data and information. For instance, States and Tribes may need a high level of confidence in an assessment, such as that associated with a Level 3 or 4 bioassessment, to determine level of stress along a gradient (Figure C-1). Less rigorous methods would not reliably detect disturbance.

Figure C-2 is a conceptual illustration depicting how increasingly comprehensive bioassessments better detect and discriminate differences along the BCG. As currently defined, Level 4 employs numeric biocriteria, based on calibrated and refined assessment tools (e.g., calibrated indexes or model output) that, in turn, are based on regional reference conditions at a sufficiently detailed level of geographic stratification and classification of aquatic ecotypes. This approach can discriminate different condition tiers (e.g., as in the Biological Condition Gradient) within a known margin of uncertainty. Level 3 usually employs a numeric and/or narrative assessment methodology that discriminates among fewer condition categories and reflects an ordinal scale of measurement (i.e., excellent, good, fair, poor). Assessment programs that rate as Level 2 may be unable to differentiate more than two broad categories or classes of condition. This level has a large degree of uncertainty about assessing stressors, and the pass/fail boundary may reflect an under- or over- protective threshold. Level 1 functions as a general screening tool and may identify best conditions from the worst in only a very coarse sense. The uncertainty with a Level 1 rated program precludes resolution to many management questions without further monitoring and assessments.

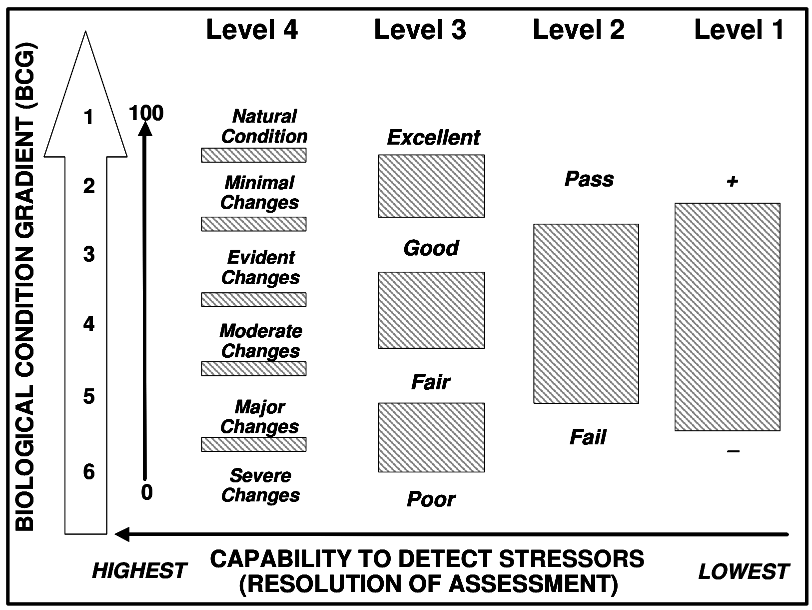


FIGURE C-2. Conceptual illustration of the capability of increasingly comprehensive bioassessments to detect and discriminate along the biological condition gradient. Shaded areas represent relative degree of uncertainty.

Appendix D

The Role of Reference Condition in Biological Assessment and Criteria

(INTRODUCTION TO DRAFT DOCUMENT ON DEVELOPMENT AND APPLICATION OF THE REFERENCE CONDITION CONCEPT)

The Clean Water Act's biological integrity objective and fishable swimmable goals pose significant challenges to States and Tribes charged with evaluating whether aquatic resources under their management achieve the objective and goals. One of the critical challenges is the development of a standard or benchmark by which to judge whether particular water bodies are in accord with the objective and goals. The concept of a "reference condition" and its implementation form the foundation on which to make such judgments.

This document provides States, Tribes, and other practitioners with guidelines on the reference condition concept and how to apply it in their water management programs, particularly for assessing the condition of aquatic resources. These guidelines are intended to broaden the implementation of biological monitoring and assessment, to increase the consistency among States and Tribes, and to improve the success of individual programs.

States, Tribes, and others have developed and implemented the concept of reference condition in a variety of ways to meet their individual needs, without comprehensive guidance from the U.S. EPA. This "bottom-up" approach has both advantages and disadvantages. Advantages include the exploration of a variety of interpretations of the concept and their implementation, yielding information on successes and difficulties. From these experiences comes an evaluation of what works and what does not. Disadvantages include the diversity of opinions about the concept and its role, leading to potential confusion and sometimes contradictory interpretation and implementation. The technical and policy challenges inherent in this effort have resulted in considerable variation in how individual States and Tribes define and use the concept. Establishing and using the reference condition concept appropriately is critical to implementing biological criteria and tiered aquatic life uses to protect and restore water resource quality. Part of the purpose of this document is to encourage consistency, both in the language that is used to express the concept, and in its everyday application.

This document will cover the following topics: a description of the concept of reference condition as well as related terms and concepts (including minimally disturbed, least disturbed, and best attainable conditions); methods for characterizing reference and related conditions; using water body classification to partition natural variability; setting thresholds to determine achievement of a target condition; and application of the concept in heavily modified regions (e.g., urban landscapes, agricultural regions) and waterbodies (reservoirs, regulated rivers). Both technical and implementation issues are addressed to increase the understanding of the concepts. A section on frequently asked questions and answers is included to address topics of particular concern to practitioners. Throughout the document, examples are drawn from existing State and Tribal programs to illustrate specific applications that are consistent with the guidelines.

In April 2003, U.S. EPA's Office of Water sponsored a National Biological Assessment and Criteria Workshop in Coeur d'Alene, Idaho. This workshop contained sessions on a variety of related topics including sessions on the reference condition concept, water quality standards, biocriteria, tiered aquatic life uses, and index development. A CD that contains many of the presentations at this workshop is included as an appendix to provide a snapshot of the state of the science at that time, and a means to flesh out some of the issues not addressed in detail in the body of this document. Material in this document supersedes any contradictory material presented on the CD because thinking has evolved since that time.

Technical and implementation issues

A principal technical challenge facing States and Tribes is accurately determining a reference or related condition from the range of historical and current ecological conditions. This may involve the analysis of data from existing reference sites and/or the modeling of historical information and expert opinion. Technical issues include understanding and taking into account natural variability through classification and/or modeling of natural gradients. Both classification and modeling need to be ecologically valid, yet practical for States and Tribes. A related issue is determining whether an existing condition is significantly (both ecologically and statistically) different from a specified condition (e.g., as specified in water quality standards). Scientific rigor is necessary, tempered by ease of understanding and implementation.

Implementation issues revolve around how States and Tribes can apply the reference condition concept to protect and improve an existing biological condition through application in water quality standards, including:

- Refinement of aquatic life uses through a) setting condition thresholds, b) interpreting/translating narrative aquatic life uses, and c) establishing subcategories of tiered aquatic life uses;
- Establishment of numeric biological criteria;
- Quantitative biological description of existing designated uses through bioassessments; and
- Determination of departure of existing condition from biological integrity.

U.S. EPA guidance on the implementation of the reference condition concept balances the need for scientific rigor and the need for practical application, that together result in the protection and improvement of water quality.

Appendix E

Statistical Guidance for Developing Indicators for Rivers and Streams: A Guide for Constructing Multimetric and Multivariate Predictive Bioassessment Models

(SUMMARY OF DRAFT DOCUMENT)

[This document has undergone various levels of review by a technical workgroup and U.S. EPA representatives. The current version is being prepared for a more comprehensive review prior to finalization. The final document is anticipated in 2005.]

States are faced with the challenges of not only developing tools that are both appropriate and cost-effective (Barbour 1997), but also the ability to translate scientific data for making sound management decisions regarding water resources. The approach to analysis of biological (and other ecological) data should be straightforward to facilitate a translation for management application. This is not meant to reduce the rigor of data analysis but to ensure its place in making crucial decisions regarding the protection, mitigation, and management of the nation's aquatic resources. In fact, biological monitoring should combine biological insight with statistical power (Karr 1987). Karr and Chu (1999) state that knowledge of regional biology and natural history (not a search for statistical relationships and significance) should drive both sampling design and analytical protocol.

A central premise of biological assessment is comparison of the biological resources of a waterbody to an expected reference condition. The condition of the waterbody is evaluated by its departure from the expected condition. Biological assessment of waterbodies depends on our ability to define, measure, and compare an assessment endpoint between similar systems. This guidance outlines analytical methodologies to perform two tasks:

- Characterize biological expectation.
- Determine whether a site deviates from that expectation.

The methods considered here use the same general approach: sites are assessed by comparing the assemblage of organisms found at a site to an expectation derived from observations of many relatively undisturbed reference sites. The expectations are modified by classifying the reference sites to account for natural variability. Biological variables are tested for response to stressors by comparison of undisturbed or minimally disturbed reference sites and disturbed sites. A set of "rules" is developed from this information, which are then used to determine if the biota of a site deviate from the expectation, indicating the degree to which the site is impacted.

Several analytical methods have been developed to assess the condition of water resources from biological data, beginning with the saprobien system in the early 20th century to present-day development of biological markers (Cairns and Pratt 1993). This document provides guidance for two methods for analyzing and assessing waterbody condition from assemblage and community-level biological information:

1. Multimetric assessment using an index that is the sum of several metrics. This is the basis of the Index of Biotic Integrity (IBI) (Karr et al. 1986), the Invertebrate Community Index (ICI) (Ohio EPA 1990); the Rapid Bioassessment Protocol (Plafkin et al. 1989); and State indexes developed from these (e.g., Southerland and Stribling 1995, Barbour et al. 1996a, Barbour et al. 1996b).

2. Assessment comparing actual species composition at a site to an idealized reference site predicted from a multivariate statistical model. This is the basis of the River InVertebrate Prediction And Classification System (RIVPACS; Wright et al. 1984, Furse et al. 1984, Moss et al. 1987, Wright 1995, Wright 2000) and the AUStralian RIVer Assessment System (AUSRIVAS; Davies 2000, Simpson and Norris 2000).

Many other methods are possible, as well as permutations of the two methods above, all of which are beyond the scope of this document. The two approaches were selected because:

- They use community and assemblage data.
- The methods are not restricted to any one assemblage. The examples all use freshwater benthic macroinvertebrates, but any other assemblage could also be used, such as fish phytoplankton, zooplankton or macrophytes.
- The methods are general, and have been used by several agencies in many areas. The examples used to illustrate the methods have also been carried out over wide geographic areas with many sites, demonstrating the generality of the methods.
- The methods have been fully documented and illustrated with case examples.
- These analysis methodologies are cost-effective and easy to communicate to managers and the public.

Once the framework for bioassessment is in place, conducting bioassessments becomes relatively straightforward. Either a targeted design that focuses on site-specific problems or a probability-based design, which has a component of randomness and is appropriate for 305(b), area-wide, and watershed monitoring, can be done efficiently. Routine monitoring of reference sites may be based on a probability design, which will allow cost efficiencies in sampling while monitoring the status of the reference condition of a State's streams. Potential reference sites of each stream class would be randomly selected for sampling, so that an unbiased estimate of reference condition can be developed. A randomized subset of reference sites can be resampled at some regular interval (e.g., a 4-year cycle) to provide information on trends in reference sites.

This document outlines the steps required to complete multimetric and multivariate predictive assessment models. It includes sections briefly covering the conceptual principles behind each step and then uses an example dataset that demonstrates the practical application of those principles step by step. It begins with a discussion of some concepts and approaches common to both techniques and then moves into multimetric and multivariate predictive models. At the end, it concludes with a discussion of how biocriteria can be developed from either of the approaches.

United States Environmental Protection Agency
Office of Water
Washington, DC 20460
(4304T)

EPA-822-R-05-001
August 2005

Important Concepts and Elements of an Adequate State Watershed Monitoring and Assessment Program

August 8, 1997

prepared by

Chris O. Yoder
State of Ohio Environmental Protection Agency
Division of Surface Water
1685 Westbelt Drive
Columbus, Ohio 43228

prepared for

U.S. EPA, Office of Water
(Cooperative Agreement CX 825484-01-0)

and

ASIWPCA Standards and Monitoring Task Force

Important Concepts and Elements of an Adequate State Watershed Monitoring and Assessment Program

I. INTRODUCTION

Watershed-based approaches are gaining widespread acceptance as a conceptual framework from within which water quality management programs should function. However, overall reductions and inequities in State ambient monitoring and assessment programs jeopardize the scientific integrity of watershed-based approaches. This also has had the undesirable effect of failing to properly equip the States and EPA to adequately meet the challenges posed by recently emerging issues such as cumulative effects, nonpoint sources, habitat degradation, and interdisciplinary issues (*e.g.,* TMDLs) in general. Unfortunately, the chronic shortfall in ambient monitoring and assessment resources is not new - the ITFM (1995) reported that of the funding allocated by state and federal agencies to water quality management activities, only 0.2% was devoted to ambient monitoring. As the need for adequate supplies of clean water increases, concerns about public health and the environment escalate, and geographically targeted watershed-based approaches increase, the demands on the water quality monitoring "infrastructure" will likewise increase. These demands cannot be met effectively nor economically without fundamentally changing our attitudes towards ambient monitoring (ITFM 1995). An adequate ambient monitoring and assessment framework is needed to ensure not only a good science-based foundation for watershed-based approaches, but water quality management in general. This paper attempts to describe the important elements, processes, and frameworks which need to be included as part of an adequate State monitoring and assessment program and how this should be used to support the overall water quality management process. Furthermore, it is a goal of this effort to highlight the need to revitalize monitoring, assessment, and environmental indicators as an integral part of the overall water quality management process.

Monitoring and assessment information, when based on a sufficiently comprehensive and rigorous system of environmental indicators, is integral to protecting human health, preserving and restoring ecosystem integrity, and sustaining a viable economy. Such a strategy is intended to achieve a better return on public and private investments in environmental protection and natural resources management. In short, more and better monitoring and assessment information is needed to answer the fundamental questions that have been repeatedly asked about the condition of our water resources and shape the strategies needed to deal with both existing and emerging problems within the context of watershed-based management.

The long-term vision is to develop a process for the comprehensive assessment of the waters of each State by producing and implementing a multi-year monitoring and assessment framework at relevant geographic scales to support all water quality management objectives (including risk-based decision making). Some of the key elements of this approach are:

- development and implementation of a statewide monitoring strategy.
- publishing existing monitoring and assessment results from all relevant sources (*e.g.,* Watershed specific reports, State 305[b] reports).
- performance of data storage, retrieval, and management.
- taking appropriate regulatory and management actions based on those results.

These efforts would fall short if a linkage between program management and monitoring and assessment were not made part of the overall water quality management process (Figure 1). This, too, is part of the long range vision for revitalizing the role of water quality monitoring nationwide.

II. GOALS OF AN ADEQUATE STATE MONITORING AND ASSESSMENT PROGRAM

The following is a compilation of the major program goals that should shape the design of an adequate State monitoring and assessment program and thus become the identifiable characteristics. While much of this is patterned after the major monitoring and assessment compendia and program guidance that has recently been developed (ITFM 1995; U.S. EPA 106 Program Guidance), the specifics of implementation lie within the custodial responsibilities of State water quality management programs.

1. The **18 national water indicators** and the goals each measures (U.S. EPA 1995a; see inset p. 3) are employed as the core indicators with additional area and/or resource specific goals and indicators as needed to fulfill the following purposes:

    - conserve and enhance public health.
    - conserve and enhance ecosystems.
    - support uses designated by States/Tribes in Water Quality Standards (WQS).
    - conserve and improve ambient conditions.
    - reduce or prevent loadings and other stressors (*e.g.*, habitat degradation).

Taken together, all of the above should lead to achieving healthy watersheds.

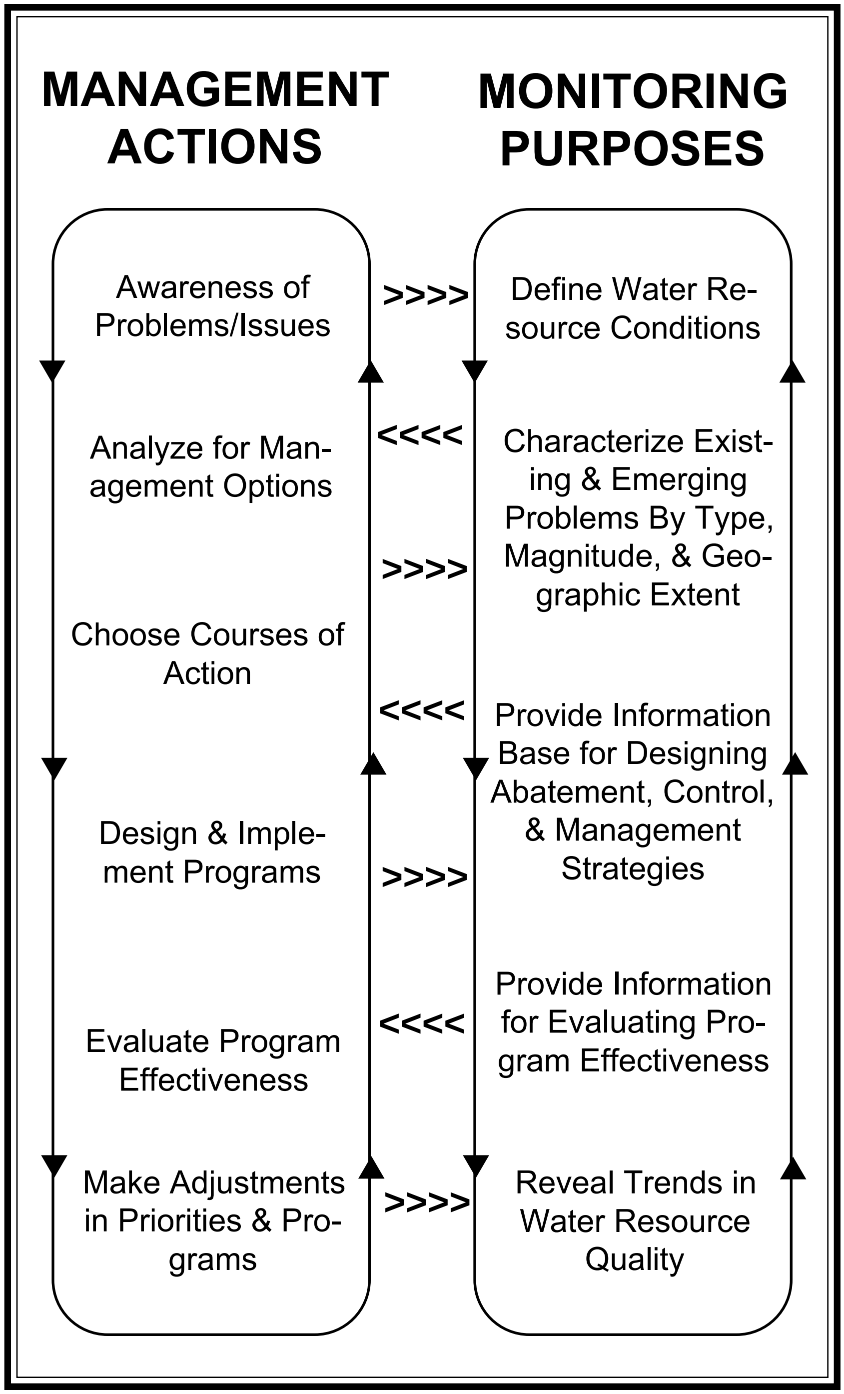


Figure 1. The relationship between management actions and the purposes monitoring and assessment (after ITFM 1995).

2. **Assess all water resource types** within an organized time frame (*e.g.*, rotating basin approach) by employing the following approaches:

The U.S. EPA National Indicators for Water and the Goals Each Supports

Conserve & Enhance Public Health:

1. Population served by drinking water systems in compliance with health-based standards.
2. Population served by drinking water systems at risk from microbial contamination.
3. Population served by drinking water systems exceeding lead action levels.
4. Number of drinking water systems with source water protection.
5. Percentage of waters with fish consumption advisories.
6. Percentage of estuarine and shellfish waters approved for harvest for human consumption.

Conserve & Enhance Ecosystems:

7. Percentage of waters with healthy aquatic communities (*i.e.*, biological integrity).
8. Percentage of imperiled aquatic species.
9. Rate of wetland acreage loss.

Support Designated Uses:

10. Percentage of waters meeting designated uses:
    a. Drinking water supply
    b. Fish and shellfish consumption
    c. Recreational
    d. Aquatic life

Conserve & Improve Ambient Conditions:

11. Population exposed to chemical pollutants in ground water.
12. Trends in surface water pollutants.
13. Concentrations of selected pollutants in shellfish.
14. Trends in estuarine eutrophication.
15. Percentage of waters with chemically contaminated sediments.

Reduce Loadings & Prevent Other Stressors:

16. Point source loadings to surface and ground water.
17. Nonpoint source loadings to surface and ground water.
18. Marine debris.

- achieve virtually 100% coverage through a mix of different spatial schemes, *i.e.*, targeted sites, rotating basin cycles, and/or probabilistic design.
- utilize appropriate and robust techniques for extrapolation and stratification of monitoring and assessment results (*i.e.*, every mile of every stream need not be monitored to achieve the 100% coverage goal).
- maximize interagency and inter-organizational cooperation and collaboration.
- when appropriate, make use of volunteer organization results.

3. Produce a **"better" 305b report:**

- national statistics are currently biased by wide differences between State approaches to monitoring & assessment including indicators usage and calibration - one result is widely divergent state estimates of impaired waters (generally overly optimistic estimates of the full attainment of aquatic life uses).
- assignment of impairment (or lack thereof) to associated causes and sources also reveals the inconsistent usage of indicators and indicator frameworks - *e.g.*, habitat has been under reported by most states (almost one-half of states reported *zero* impaired miles for rivers & streams in 1992).

4. Support the emerging **watershed approaches**:

- reductions in State monitoring & assessment programs jeopardize the science basis for successfully implementing watershed-based approaches which are ostensibly based (in part) on addressing previously overlooked or under-emphasized problems.
- management applications most commonly take place at the watershed level thus monitoring & assessment must be relevant to this level of management and be capable of detecting impairments and characterizing aquatic resources at this scale.

5. **Satisfy basic questions** that are frequently encountered by water quality program managers:
   - what is the condition of surface, ground, estuarine, and coastal waters?
   - how and why are conditions changing over time?
   - what are the associated causes and sources of impairment?
   - are water quality management programs producing the desired results?
   - are state and national water quality goals being attained?

Each of the above can be subdivided into issue specific questions that are commonly encountered by water quality managers (see inset at right).

6. **Integrate the water resource integrity concepts** that have been developed during the past 10-15 years into monitoring and assessment approaches, environmental indicators, and watershed-based programs:

   - the five factors that determine the integrity of water resources (Figure 2; Karr *et al.* 1986) should be used to guide the development of environmental indicators - indicators which both represent or extend to each major factor *and* which reflect the integrity of the water resource as a whole (*e.g.*, composite measures, indices) are needed.
   - follow the stressor, exposure, response paradigm for determining the most appropriate roles for individual indicators - *avoid the inappropriate substitution of stressor and exposure indicators for response indicators.*
   - utilize appropriate regionalization schemes (*e.g.*, ecoregions, subregions) to stratify and partition natural variability for ambient indicators.
   - incorporate tiered and refined use designations in the State WQS as appropriate.
   - use the water indicators hierarchy (Figure 3) as an operational framework for State water quality management programs - make linkages between administrative activities and indicators of stress, exposure, and response.

Water Quality-Based Decisions Which Would Benefit From Better Monitoring & Assessment Information

Water Quality Standards:
- Refined and stratified designated uses and criteria
- Biological criteria
- Site-specific applications (*e.g.*, dissolved metals translators, design temperature & pH, hardness)
- Water effect ratios
- Anitdegradation
- Ground truthing revisions to water quality criteria

TMDLs:
- Delineating impaired segments and associated causes & sources
- Wasteload allocation (model calibration & verification

NPDES Permits:
- Impact assessment
- Toxicity assessment (*i.e.*, WET testing)
- Overall permit program effectiveness

Nonpoint Sources:
- Delineating impaired segments and prioritization of watersheds
- Database for State Nonpoint Source Assessments

404/401 Dredge & Fill:
- Improved site-specific review and approval criteria
- Minimize exemptions via nationwide permits

Ground Water:
- Development of ambient background characteristics

Wetlands:
- Improved wetlands classification and delineation criteria

III. STATE MONITORING & ASSESSMENT PROGRAM OBJECTIVES

The following are some of the major objectives that State monitoring & assessment programs should have as priorities. Fully meeting some of these objectives will require time to acquire and develop the necessary database, indicators, and staff expertise. However, this will be partly dependent on the status of existing and past State monitoring and assessment efforts. Nevertheless, using the following objectives provides a basis for determining the adequacy of a given State program. A well rounded approach to indicators and monitoring design utilizing a core set of chemical, physical, and biological indicators should provide the information needed to simultaneously meet these objectives without the need to redesign the approach for each different objective.

1. Baseline characterizations of surface water resources:

- status and trends information.
- aquatic resource characterization.

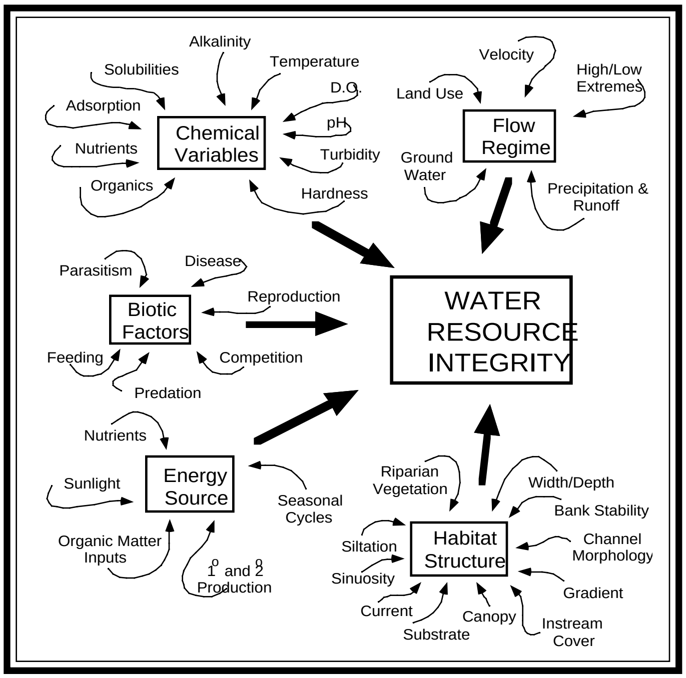


Figure 2. The five major factors which determine the integrity of the water resource (modified after Karr et al. 1986).

2. Identification and characterization of existing and emerging problems:

- selection of indicators and the overall indicator framework will strongly influence the adequacy of problem identification and characterization (we cannot address problems that we do not know about or adequately understand).
- the indicator framework and monitoring design must be prepared to provide information and insights to problems that may not yet be understood or even recognized.
- there will be a need to go beyond point source paradigms.
- make better linkages between designated uses and indicators.

3. Guide and evaluate the water quality management and regulatory process:

- monitoring & assessment information should drive the regulatory and management processes from problem identification to assessing the effectiveness of these efforts.
- the 305[b] process (*i.e.,* Water Body System) should be the central reporting mechanism for State programs - this will further benefit the national assessments compiled by EPA, other federal agencies, and private organizations.
- support the development and refinement of aquatic life and other designated uses in State WQS.
- examples of other regulatory and management programs that can be influenced include 303[d] listing, TMDLs, water quality-based permitting, compliance and enforcement, prioritizing grants and other financial assistance, the State nonpoint source assessment (319 program), etc.
- monitoring and assessment information should provide the impetus for “new” regulatory or program management directions (*e.g.,* initiatives to restore and protect riparian habitat, nutrient criteria, sediment criteria, stream protection, antidegradation) and enhance existing efforts (CSOs, stormwater, 404/401 program, chemical criteria validation, biological criteria).

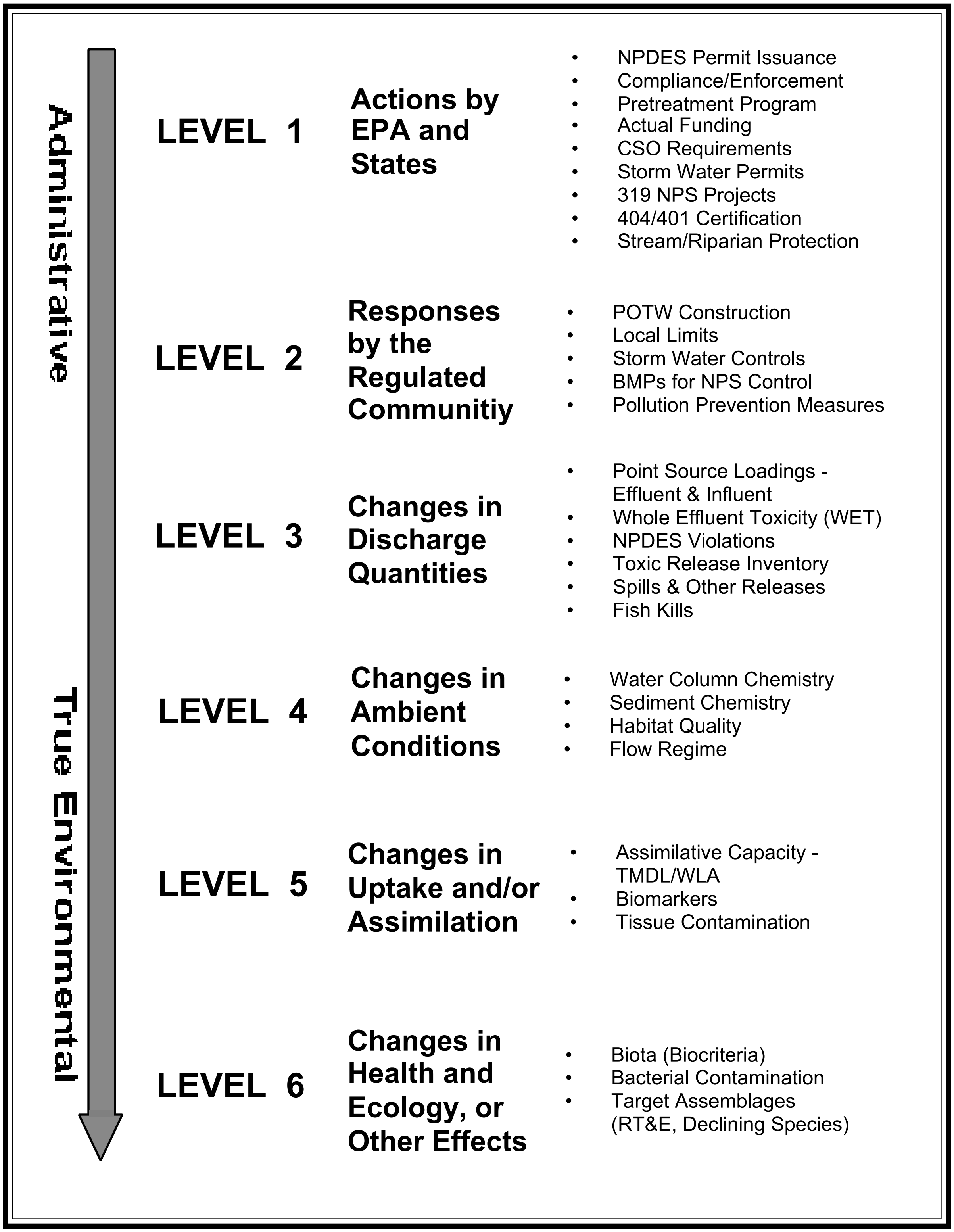


Figure 3. Hierarchy of administrative and environmental indicators which can be used by States for monitoring and assessment, reporting, and evaluating overall program effectiveness. This is patterned after a model developed by U.S. EPA (1995b).

4. Evaluation of overall water quality management program effectiveness:

   - demonstrate the effectiveness of 25+ years of CWA program implementation.
   - establish linkages between administrative activities (*i.e.,* “bean counts”) and environmental results (*i.e.,* ambient chemical, physical, and biological indicators).
   - which actions worked and which ones did not? - provide insights on why and suggest what specific program and/or resource adjustments might be needed.

5. Responding to emergencies, complaint investigations:

   - quantify environmental damages on a spatial and/or temporal basis.
   - characterize resources at risk.
   - define the magnitude of apparent problems.

6. Identify and characterize reference conditions:

   - baseline for development of indicator benchmarks for evaluating designated use attainment/ non-attainment (*e.g.,* biological criteria) and other management objectives.
   - this functions as a long term data source for characterizing ambient biological, chemical, and physical conditions through time.

IV. MONITORING & ASSESSMENT PROGRAM DESIGN ISSUES

Monitoring and assessment program design includes the different types of indicators and the frameworks within which each is developed and used. This in turn determines the different types of data that will need to be collected and synthesized into information in order to successfully realize the previously stated goals and objectives. Spatial considerations about the basic design of the monitoring program are also included and will be most influenced by the overall program goals and objectives of each State. State monitoring and assessment programs serve multiple needs and must function across multiple scales (*i.e.,* local watershed, basin/subbasin, statewide), thus consideration of more than one approach will likely be needed.

Environmental Indicators for Surface Waters

1. The most appropriate roles of indicators are defined as follows:

   - Stressor Indicator - measures of activities which have the potential to impact the environment (*e.g.,* pollutant loadings, land use characteristics, habitat changes).
   - Exposure Indicator - measures of change in environmental variables which suggest a degree (magnitude and duration) of exposure to a stressor (*e.g.,* chemical pollutant levels in water and sediment, toxicity response levels, habitat quality indices, biomarkers).
   - Response Indicator - usually a composite measure or other expression of an integrated or cumulative response to exposure and stress (*e.g.,* biological community indices, status of a target species, etc.).
   - The problem nationally with inconsistent 305[b] statistics (and by extension inconsistent 303[d] and 304[l] lists, etc.) is usually the result of the inappropriate substitution of stressor and/or exposure indicators in the place of response indicators - this is commonly due to the lack of

information about response indicators.

- The exclusion of response indicators and the inappropriate substitution with exposure and/or stressor indicators ultimately influences what States report in terms of waters meeting designated uses. An example of this is illustrated in Figure 4 where some State estimates of aquatic life use attainment based on surrogate approaches are much different than estimates based primarily on biological assessments (U.S. EPA 1996).

2. Use the EPA hierarchy of indicators (U.S. EPA 1995b; Figure 3) as a template to improve the integration of administrative actions and measures with environmental indicators within the State water quality management process:

   - The EPA hierarchy of surface water indicators links traditional administrative approaches (permitting, funding, compliance, enforcement) with environmental indicators which simultaneously sequences stressor, exposure and response indicators - six levels (Figure 3).
   - The six level hierarchy can become an operational template for implementing environmental indicators and monitoring information within a State water quality management process via a watershed approach. This will facilitate the development of case histories about what works and what does not, showing where information gaps exist, and providing opportunities for feedback throughout the process. An example from the Ohio pilot water indicators demonstration project is included in the selected examples (Part IX.).

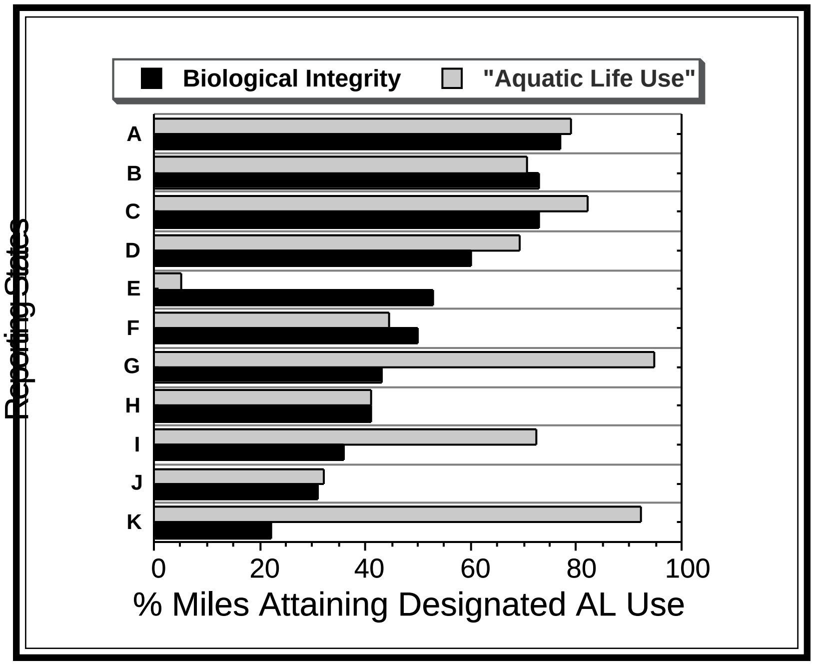


Figure 4. Miles of rivers and streams reported as fully supporting designated aquatic life uses based on varying methods used by 11 states in their 305[b] reports (light shading) compared to that based on biological assessments (after U.S. EPA 1996).

Monitoring Design Approaches

A key issue facing the States and EPA is selection of an appropriate monitoring design. It has been recognized for some time that the traditional fixed station design (*e.g.,* NAWQMN, NASQAN) common to many State monitoring networks is alone insufficient to meet the above stated objectives. However, State monitoring and assessment resources even under the best of circumstances have been limited and therefore must be prioritized. Thus, selection of the most cost and information effective spatial design is a critical step in the process. Two approaches, a synoptic, targeted design commonly referred to as a rotating basin approach and the probabilistic design developed by the U.S. EPA EMAP program are summarized here. The strengths and weaknesses of each are indicated with respect to the multiple issues that State monitoring and assessment programs must address. A case example from the Ohio portion of the E. Corn Belt Plains ecoregion Regional EMAP project is included in Part IX.

Rotating Basin Approach

1. Strengths:
   - organized, systematic approach based on accumulating assessment information at a local scale over a fixed period of time, usually 5 or 10 years.
   - coincides with various management programs which are supported by the monitoring & assessment information (*i.e.,* NPDES permit reissuance, basin-wide water quality planning, proposed 5-year 305b reporting cycle).
   - provides monitoring & assessment information at a local or reach specific scale so that the many issues which occur at this level can be addressed while providing the opportunity to aggregate upwards to a watershed, regional, statewide, or national scale once sufficient data exists.
   - there is more opportunity to define gradients of specific human disturbances with assessment information (*e.g.,* Karr's human activity "dose" - ecological response curve).
   - develop and maintain tabs on reference condition in a predictable and standardized time frame.

2. Weaknesses:
   - visiting a basin/segment/watershed only once in 5 or 10 years may not be sufficient to satisfy all needs.
   - larger scale assessment information (*i.e.,* in support of a valid statewide assessment) is generally not available for 5-10 years.

Probabilistic Design

1. Strengths:
   - statistically robust design.
   - "faster" route to a statewide assessment - aggregate to national scale.
   - transcends State boundary limitations - can facilitate collaborative monitoring between States.

2. Weaknesses:
   - lacks site-specific/issue-specific resolution.
   - logistics are potentially more difficult (*i.e.,* more difficult access to remote monitoring sites).
   - reference condition may be more difficult to define on probability basis alone.
   - local scale issues may be overlooked.

V. AQUATIC RESOURCE CHARACTERIZATION

Defining the different aquatic resource types that a State program must address is a critical step in the process. This includes the major aquatic ecosystem types such as flowing waters (*i.e.,* rivers and streams), lakes and reservoirs, coastal waters, great lakes, estuaries, or wetlands. Further stratification within each is possible (*e.g.,* headwater streams, wadable streams, large rivers, depressional wetlands, riparian wetlands, etc.) and may be accounted for *a priori* or as part of the indicator development and calibration process. Other stratification elements, which includes watershed driving factors (*e.g.,* ecoregions) and other physical vectors, are incorporated as well. Designated aquatic life uses provide an additional layer of stratification. Taken together all of these processes should result in more finely tuned indicator expectations or benchmarks against which management program success will ultimately be judged.

VI. STATE MONITORING & ASSESSMENT COMPONENTS AND RESOURCES

State monitoring and assessment programs need to include the appropriate ambient measurements in order to adequately meet the previously stated goals and objectives. The Intergovernmental Task Force on Monitoring Water Quality (ITFM 1995) recommended the minimum elements of an adequate monitoring and assessment program that will support meeting the previously stated goals and objectives (Table 1). This also represents the elements essential to implementing the hierarchy of water indicators framework (Figure 3) which, in turn, is needed to not only demonstrate program effectiveness, but provide opportunities for feedback resulting in future program improvements.

The ITFM (1995) concluded that the implementation of the ITFM recommendations and strategy would result in an adequate information base to achieve the environmental protection and natural resource management goals and objectives established for the nation's aquatic resources. However, it was also recognized that full implementation of the strategy could not be achieved "overnight" and that the necessary capacity and resources (*i.e.,* the monitoring and assessment "infrastructure") will need to be acquired over a reasonable period of time. Nevertheless, monitoring organizations, including States, will need to review, update, and/or revise their monitoring strategies in a series of deliberate steps. The demands that are increasingly being placed on our water resources at all scales require that past approaches to monitoring be significantly improved both in terms of quality and quantity. Some of the steps towards a more comprehensive and effective approach to ambient monitoring include the following which also summarizes the major points of this document:

1. Develop a goal oriented approach to monitoring, assessment, and indicators development where indicators are sufficiently specific so as to explicitly measure the identified national goals and those relevant to State WQS.

2. Evaluate information priorities and identify existing information gaps.

3. Develop a comprehensive and flexible approach that addresses all relevant scales and aquatic resource types.

4. Take advantage of inter-organizational collaboration whenever appropriate.

5. Link traditional compliance monitoring with watershed-based ambient monitoring.

6. Deal effectively with methods comparability to maximize the flexibility in monitoring and assessment approaches while producing data and information of known quality and power of assessment.

7. Automate and streamline data and information management including data entry, storage, and retrieval.

8. Develop better assessment and reporting at all relevant scales; publish results on a regular basis.

9. Promote the development of incentives and the elimination of disincentives to the development of better State ambient monitoring programs and indicators.

Table 1. Summary matrix of recommended environmental indicators for meeting management objectives for status and trends of surface waters (shaded boxes with ***X*** *are recommended as a primary indicator after ITFM 1995; other recommended indicators are indicated by* **✓).** *The corresponding EPA indicator hierarchy level is also listed between indicator groups.*

	Categories of Management Objectives					
	Human Health		**Ecological Health**		**Economic Concerns**	
Indicator Group	Consumption of Fish /Shellfish	Public Water Supply	Recreation (swimming, fishing, boating)	Aquatic/ Semi-aquatic Life	Industry/ Energy/ Transportation	Agriculture/ Forestry
Biological Response Indicator (Level 6)						
Macroinvertebrates		X	X	X		X
Fish	X		X	X		X
Semiaquatic Animals	X		X	X		X
Pathogens	X		X			X
Phytoplankton	X	X	X	X	X	
Periphyton				X		
Aquatic Plants		X	X	X	X	X
Zooplankton		X	X	X		X
Chemical Exposure Indicator (Level 4&5)						
Water chemistry	X	X	X	X	X	X
Odor/Taste	X	X	X			
Sediment Chemistry	X	X	X	X	X	X
Tissue Chemistry	X	X	✓	X	X	
Biochemical Markers	✓	✓	✓	✓		✓
Physical Habitat/Hydrologic Indicator (Levels 3&4)						
Hydrological Measures	X	X	X	X	X	X
Temperature	X	X	X	X	X	✓
Geomorphology	X	X	X	X	X	X
Riparian/shoreline	X	X	✓	X	X	X
Ambient Habitat Quality	✓	✓	✓	✓	✓	✓
Watershed Scale Stressor Indicators (Levels 3,4&5)						
Land Use Patterns	X	X	X	X	X	X
Human Alterations	X	X	X	X	X	✓
Watershed Impermeability	✓	✓	✓	✓	✓	✓
Pollutant Loadings Stressors (Level 3)						
Point Source Loadings	✓	✓	✓	✓	✓	✓
Nonpoint Source Loadings	✓	✓	✓	✓	✓	✓
Spills/Other Releases	✓	✓	✓	✓	✓	✓

Simply upgrading the monitoring program to include more and better measurements and the better conversion of data to information, while important, is alone insufficient. To achieve the overall goal of improving the use of monitoring and assessment information in the emerging watershed approach, water quality management must mature to focus primarily on the condition of the environment as the overall measure of program success (Figure 5). Whereas the performance of the "program" was

Two Approaches to Watershed-Based Water Quality Management

	PROGRAM FOCUSED APPROACH	RESOURCE FOCUSED APPROACH
Goal:	Program Performance	Environmental Performance
Measures:	Administrative Actions	Indicator End-points
Results:	Improve Programs	Programs are Tools to Improve the Environment

Figure 5. The goals, measures, and results of program based and resource based approaches to water quality management. State programs will evolve towards a resource based approach by developing and using a sufficiently comprehensive and rigorous system of environmental indicators.

once the principal measure of effectiveness, the program must be viewed as a tool to be used along-side monitoring and assessment and environmental indicators to improve the quality of the environ-ment.

VIII. REFERENCES

ITFM (Intergovernmental Task Force on Monitoring Water Quality). 1995. The strategy for improving water-quality monitoring in the United States. Final report of the Intergovernmental Task Force on Monitoring Water Quality. Interagency Advisory Committee on Water Data, Washington, D.C. + Appendices.

Karr, J. R., K. D. Fausch, P. L. Angermier, 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 5: 28 pp.

U.S. Environmental Protection Agency. 1996. Summary of state biological assessment programs for streams and rivers. EPA 230-R-96-007. U. S. EPA, Office of Policy, Planning, & Evaluation, Washington, DC 20460.

U.S. Environmental Protection Agency. 1995a. Environmental indicators of water quality in the United States. EPA 841-R-96-002. Office of Water, Washington, DC 20460. 25 pp.

U.S. Environmental Protection Agency. 1995b. A conceptual framework to support development and use of environmental information in decision-making. EPA 239-R-95 012. Office of Policy, Planning, and Evaluation, Washington, DC 20460. 43 pp.

IX. INDICATORS & PARAMETERS FOR ADEQUATE STATE MONITORING & ASSESSMENT PROGRAMS

The following supplemental figure shows core and supplemental indicators and parameters that are used in an adequate State monitoring and assessment program. This is patterned after the recommendations of the Intergovermental Task Force on Monitoring Water Quality (ITFM 1995). The core indicators are measured everywhere and are supplemented by a variety of chemical and physical measurments depending on the applicable designated use(s) and watershed-specific needs.

CORE INDICATORS/PARAMETERS

• Fish Assemblage • Macroinvertebrates • Periphyton

(Use Community Level Data From At Least Two)

Physical Habitat Indicators

- Channel morphology • Flow
- Substrate Quality • Riparian

Chemical Quality Indicators

- pH • Temperature
- Conductivity • Dissolved O_2

For Specific Designated Uses Add the Following Parameters:

AQUATIC LIFE

Base List

- Ionic strength
- Nutrients, sediment

Supplemental List

- Metals (water/sediment)
- Organics (water/sediment)

RECREATIONAL

Base List

- Fecal bacteria
- Ionic strength

Supplemental List

- Other pathogens
- Organics (water/sediment)

WATER SUPPLY

Base List

- Fecal bacteria
- Ionic strength
- Nutrients, sediment

Supplemental List

- Metals (water/sediment)
- Organics (water/sediment)
- Other pathogens

HUMAN/WILDLIFE CONSUMPTION

Base List:

- Metals (in tissues)
- Organics (in tissues)

Supplemental Figure 1. Core indicators and parameters for an adequate State watershed monitoring and assessment program with supplemental chemical parameters according to the applicable designated use(s). Parameters are added based on site and watershed-specific needs and overall water quality management objectives.

X. CASE EXAMPLES
(ASIWPCA Meeting Version)

Case examples of how monitoring and assessment information based on an integrated water indicators framework can be used to address some of the key goals and objectives of this guidance document are appended**.** These examples provide tangible evidence of how good monitoring and assessment information can be used to not only support specific program areas, but the overall water quality management process in general.

A. Pennsylvania DEP
The Pennsylvania examples show how the DEP is responding to the settlement of a TMDL suit by committing to increased monitoring and assessment (biological monitoring in particular) statewide.

B. Tennessee Valley Authority (TVA)
The TVA has traditionally been a leader in using ambient monitoring information to meet their water quality management obligations. The examples appended here portray the types of monitoring and assessment, the spatial design, and how this has fostered a better approach to inter-organizational collaboration.

C. Wisconsin DNR
A published paper from the Wisconsin DNR shows how biological and habitat information was used to determine the effects of nonpoint sources and land use on the integrity of Wisconsin streams. This should begin to point out how this type of information can be used in the TMDL process.

D. Ohio EPA
A number of examples from the Ohio EPA surface water monitoring and assessment program are presented and include:

- fact sheets from the 1996 Ohio Water Resource Inventory (305b report);
- watershed profiles from two basin survey areas.
- preliminary results from the E. Corn Belt Plains Ecoregion REMAP project;
- a synopsis of figures from the pilot water indicators project; and,
- three examples of how ambient monitoring data can be used to validate and/or derive chemical water quality criteria.

E. U.S. EPA, Office of Water
The most recent version of the U.S. EPA Section 106 monitoring guidance attempts to foster helping States to achieve the many goals and objectives stated herein.

XI. OHIO EPA CASE EXAMPLES:

I. 1996 Ohio Water Resource Inventory (305[b] Report) Fact Sheets:

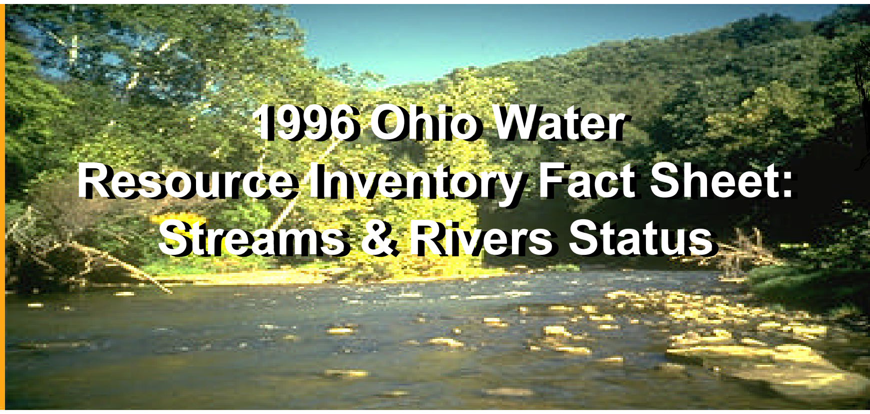
- Streams and Rivers Status
- Causes and Sources of Impairment
- Streams and Rivers: Siltation & Habitat Destruction
- Impaired Waters in Ohio: What Does This Mean?

II. An Evaluation of Spatial Monitoring & Assessment Design: Preliminary Results from the E. Corn Belt Plains REMAP Project

III. Ammonia Fact Sheets
- Associations Between the Index of Biotic Integrity and Unionized Ammonia in Ohio Rivers and Streams: A Preliminary Analysis
- Associations Between the Index of Biotic Integrity and Total Ammonia in Ohio Rivers and Streams: A Preliminary Analysis

IV. Ohio EPA Pilot Indicators Project figures

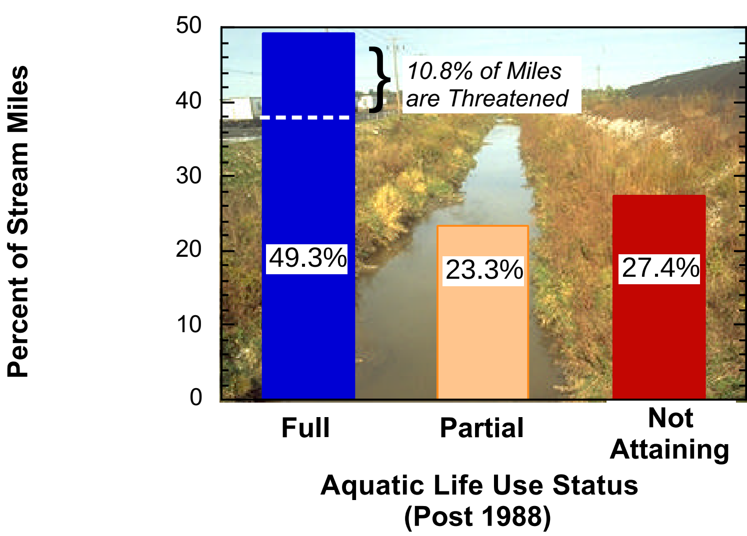
V. Watershed Profile Summaries
- Sandy Creek
- Little Miami River



1996 Ohio Water Resource Inventory Fact Sheet: Streams & Rivers Status

Ohio is a water-rich state with more than 25,000 miles of named and designated streams and rivers and a 451-mile border on the Ohio River. The suitability of these waters to support human uses (*e.g.,* recreation and drinking water) and to maintain healthy ecological conditions or "biological integrity" is critical to the sustainable future of Ohio's economy and standard of living.

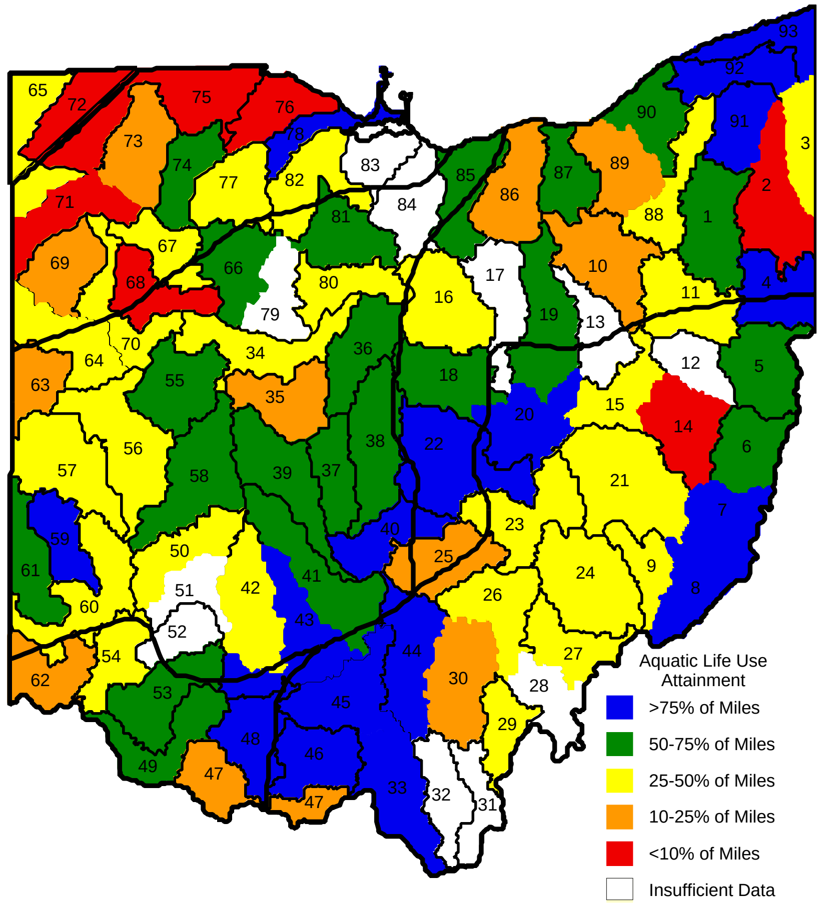
Ohio uses the fish and invertebrate communities found in streams to assess the health and well-being of Ohio's flowing waters. Aquatic animals are generally the most sensitive indicators of pollution because they inhabit the water all of the time and because of the direct contact of their gills with the water. A healthy stream community is also associated with high quality recreational opportunities (*e.g.,* fishing and otheroutdoor-related activities).



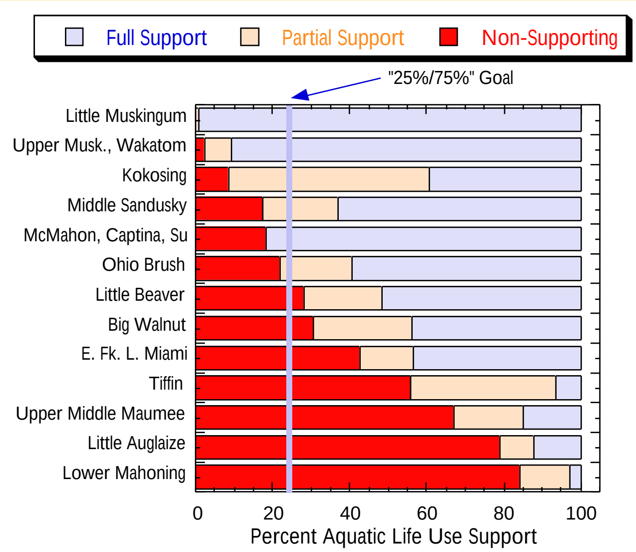
Aquatic Life Condition in Ohio's Rivers and Streams

The shoirt-term goal is for 75% of the stream and river miles to fully attain the applicable aquatic life standards (called "uses") by the year 2000. The most recent Ohio Water Resource Inventory (Ohio EPA 1996) reported that **49.3%** of streams and rivers were fully supporting the applicable aquatic life "uses". This means that nearly one-half of Ohio's streams, other than a small proportion of waters maintained as ditches or other physically limited waters, and rivers harbor **good** or **exceptional** quality fish and/or aquatic invertebrate assemblages. Streams that are considered as "partially" supporting aquatic life means that while either the fish or aquatic insects are good or excellent, the other group is only in **fair** condition. In such cases certain sensitive species may be absent or there are too many pollution tolerant species (*e.g.,* carp) than in a comparable stream where there is less pollution. "Non-attaining" streams and rivers are waterbodies in which the fish and aquatic invertebrates are both fair or one group is in **poor or very poor** condition. Examples of such streams and rivers include a warmwater stream where we should expect to find **good** fish and aquatic invertebrate communities, but both groups are rated as **fair**; or an exceptional stream, where we expect to find **exceptional** fish and invertebrates, but where both groups are **good**. As summarized in the pie chart below, the non-attainment designation does not mean that the stream is "dead," but rather represents varying degrees of unacceptable impairment.

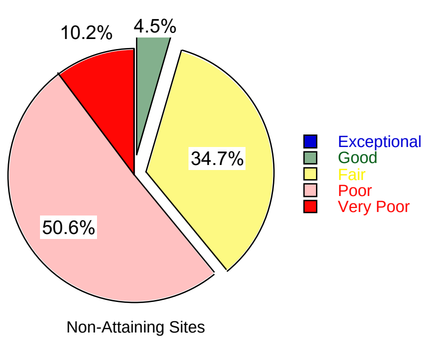
It is also helpful to look at stream and river quality from a regional perspective. The map on this page (upper right) summarizes the condition for each of 93 watersheds ("subbasins") in Ohio. Some areas of the state generally have a higher proportion of high quality streams and rivers (central and southern Ohio) than other areas (northwest Ohio). By using this perspective, we can see which watersheds are currently meeting or exceeding the Ohio 2000 goal. These will be priorities for protection. For watersheds that are far below the Ohio 2000 goal, there will be a need to evaluate the "restorability" potential and future restoration efforts prioritized. It is clear that a watershed approach, that includes efforts to restore habitat and decrease sedimentation (two of the leading causes of impairment), needs to be central to any strategy to reach the Ohio 2000 goal.



Map of Aquatic Life Use Attainment By Subbasin



Column Chart of Aquatic Life Use Attainment For Selected Subbasins



Ranges of Narrative Data in Sites Not Attaining Aquatic Life Uses

For more information contact: Ed Rankin, Division of SurfaceWater
1800 WaterMark Drive, Columbus, OH 43215-1099
(614)-728-3388; e-mail: erankin@central.epa.ohio.gov

1996 Ohio Water Resource Inventory Fact Sheet: Streams & Rivers Causes & Sources of Impairment

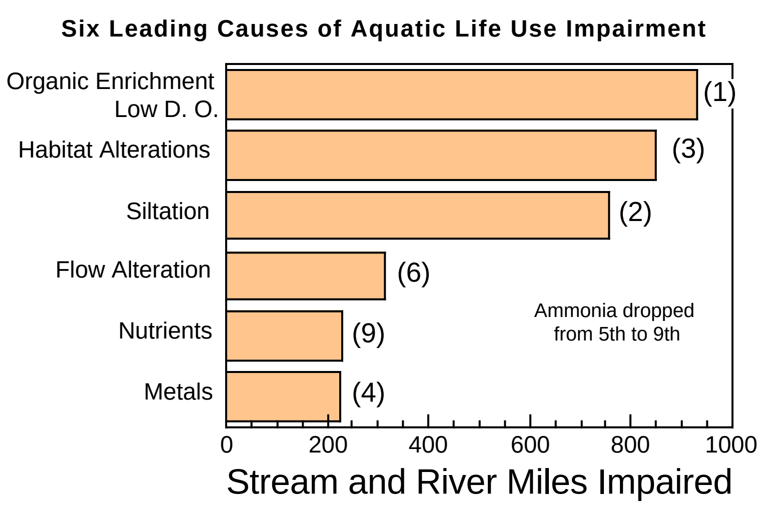
Ohio's streams and rivers have seen a substantial improvement in quality over the past 10-15 years. The majority of this improvement has been a result of investments and improvements in municipal wastewater treatment plants across Ohio.

Ohio uses the fish and invertebrate communities found in streams to assess conditions in Ohio's flowing waters. Aquatic animals are generally more sensitive to pollutants compared to other animals because they inhabit the water all of the time. A healthy stream community is also associated with higher quality recreation opportunities (e.g., fishing, canoeing, and other outdoor-related activities).

In addition to the biological data, Ohio EPA also collects information on the chemical quality of the water, sediment and effluents; data on the contaminants in fish flesh; and data on the physical nature of streams (*i.e.*, aquatic habitat, siltation). This data is essential to identify the factors that are limiting or impair aquatic life and which constitute threats to human health.

Causes of impairment are the "agents" that actually damage or impair the aquatic life in a stream, such as the toxic effects of heavy metals or acidic water. **Sources** of impairment are the origin of the agent. For example, an industry may discharge a heavy metal or a coal mine may be the source of acid water leaching into a stream.

The leading **causes** of impairment to aquatic life in Ohio streams are listed in the adjacent figure (bottom left). The leading cause is **organic enrichment**, which includes low dissolved oxygen and excessive organic pollutants. This largely originates from the inadequate treatment of municipal wastewater (a "point source") and is the most rapidly *declining* cause of impairment. **Habitat alterations** and **siltation** are the second and third leading causes and will likely emerge as the leading causes in two or three years. These causes are termed "nonpoint source" in origin because they do not emanate from pipes, but instead are a result of

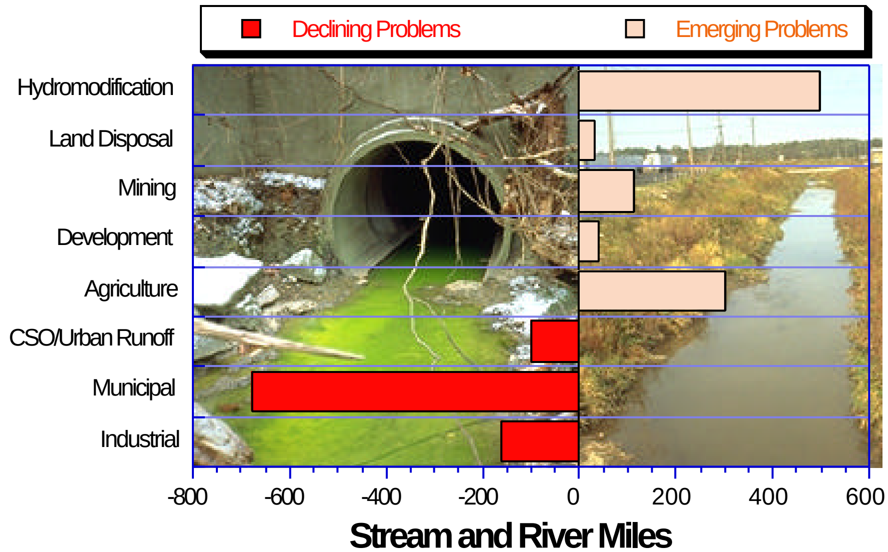


Causes of Impairment in Ohio's Rivers and Streams

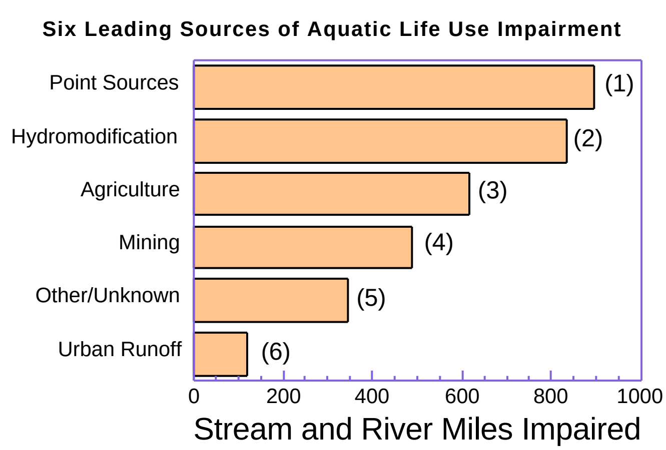
land use activities or direct disturbance of stream ecosystems (*e.g.,* by dredging, urbanization, riparian vegetation removal).

Other point source-related **causes** of aquatic life impairment have also declined in importance (see top right figure). Impacts from heavy **metals** (*e.g.,* copper, cadmium, lead, etc.) have declined from the third leading cause to the sixth since 1988. **Ammonia**, a toxic component of municipal wastewater, has dropped from the second leading cause in 1988 to ninth. This dramatic improvement resulted from the construction of new sewage treatment plants in the 1980s at a cost of approximately $6 billion thorughout Ohio.

The leading **sources** of impairment are listed in the figure below. **Point sources** of impairment are the most rapidly declining source. The importance of **hydromodification** (activities that result in habitat degradation) as a leading source of impairment will likely increase over the next several years. This trend is illustrated in the figure (see above) that compares declining and emerging sources of impairment over the past 15 years. Such mpacts are termed "emerging" problems because while always present, they were frequently masked by the more severe point source impacts of the past.



Identification of declining and emerging sources of aquatic life impairment

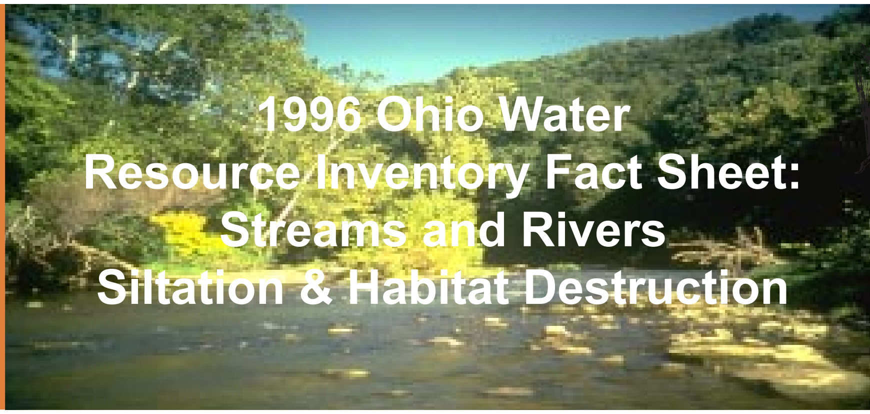


The information and knowledge illustrated in this fact sheet will be incorporated into the Ohio EPA strategic planning process, which will direct future efforts to protect and restore the water resources of Ohio in a cost-effective and scientifically sound manner.

Sources of Impairment in Ohio's Rivers and Streams

The aquatic life in a stream is a sensitive measure of the overall quality of the resource.

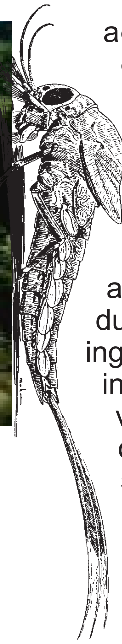
For more information contact:
Ed Rankin, Division of SurfaceWater
1800 WaterMark Drive
Columbus, OH 43215-1099
(614)-728-3388
e-mail:
erankin@central.epa.ohio.gov



1996 Ohio Water Resource Inventory Fact Sheet: Streams and Rivers Siltation & Habitat Destruction

Ohio's streams and rivers have seen a substantial improvement in the their quality over the past 10-15 years generally a result of improvements in "point sources" of pollution across Ohio. As a result of this, much of the remaining impairment to aquatic life is the result of impacts that are termed "nonpoint" sources (see Fact Sheet FS\DSW-EAS-97-3). The leading nonpoint causes that impair aquatic life are siltation and habitat modification. The causes are the result of many different nonpoint sources, especially suburban and urban development, agriculture, and flood control.

What Is "Siltation/Sedimentation?"
Siltation and sedimentation is the erosion of small particles of soil from the land surface or stream banks into the channel of a stream or river. Erosion is a natural process, however, certain human activities can greatly accelerate the rate of erosion into streams faster than the streams can export sediment downstream or expel it onto the floodplain. The impacts to aquatic life arise largely due to the the smothering of living and spawning areas. Stream surveys on Ohio have documented the loss of sensitive aquatic species, including sport species such as smallmouth bass where siltation and sedimentation is high.[1]



Sediments from eroding banks smother aquatic habitat (e.g., riffles) and fill pools where sport fishes dwell.

What Is "Habitat?"
Most aquatic species live in specific types of stream habitat. For example, sensitive species such as darters are typically found in riffles, while large predators (e.g., smallmouth bass) spend much time in pools or other deep areas (see photo in lower lefthand corner of this page). Most natural streams and rivers in Ohio have a diverse array of habitats characterized by a meandering form with numerous riffles, runs, pool, islands, gravel bars, backwaters, etc. In such natural streams that are not impacted by "point sources", stream surveys result in many species of fish(up to 40 or more) and macroinvertebrates (up to 100) in a single sample!

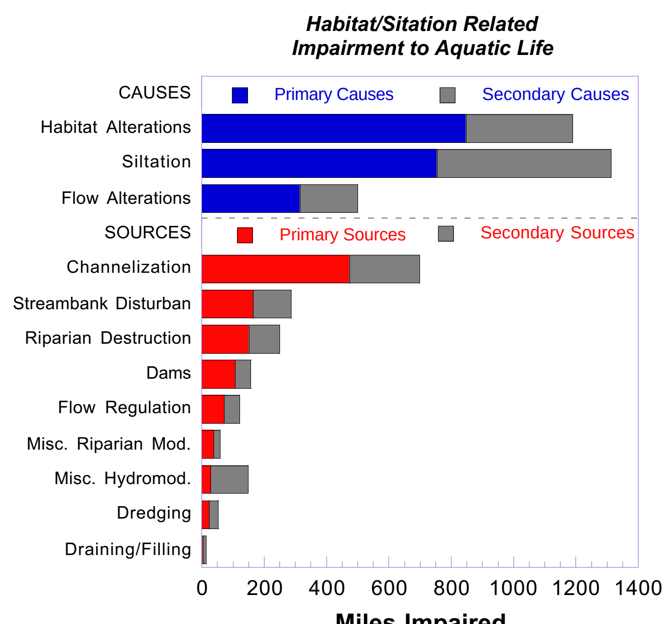
Habitat and Siltation Impacts
Most habitat and sediment impacts result from direct modification to a stream or land uses that encroach on the riparian forests along a stream. The miles of aquatic life impairment caused by habitat modification, siltation and flow alteration are listed in

Stream Habitat Types

the top of the figure below. The origin of causes include activities such as agriculture, urban/sub-urban runoff, and development related construction. Situations where the the source of the impact is directly attributed to specific hydromofication activities are listed on the figure below.

Channelization of streams for agricultural drainage or flood control is the most frequent activity that degrades habitat (see photo right). The key for protecting and and restoring aquatic habitat in Ohio is eliminating stream modifications where they are not absolutedly needed and protecting stream riparian areas from encroachment or conversion to inappropriate land uses. It is the most serious type of impact because it is essentially irretrievable, especially for our highest quality streams (Exceptional Warmwater Habitat). ODNR has experience with habitat restoration and enhancement, through an understanding of the self-stabilizing tendicies of streams, that often can serve both the environment and the need to reduce erosion and costs associated with maintaining streams in an "altered" condition. Those interested can contact the Division of Soil and Water or the Division of Wildlife at ODNR

Sources of Impairment in Ohio's Rivers and Streams



The information illustrated in this fact sheet will be incorporated into the Ohio EPA strategic planning process. This process will direct future efforts (i.e., monitoring, assessment, education and regulation) to protect and restore the waters of Ohio in a cost-effective and scientifically sound manner. Protecting stream and riparian habitat in Ohio is a key to maintaining a quality of life that Ohioians expect into the next Century.

[1]Source: Ohio Water Resource Inventory: 1996. Ohio EPA, Division of Surface Water, 1800 WaterMark Drive, Cols., Ohio 43216

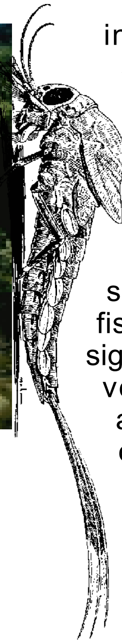
Impaired Waters in Ohio: What Does This Mean?

Positive progress has been made in improving the quality of Ohio rivers and streams. More river and stream miles meet water quality standards today (49.3%) than eight years ago (34%). This improvement is largely due to the effectiveness of efforts in reducing point sources of chemical pollution. While this progress is encouraging, one-half of river and stream miles do not meet standards. However, this does not mean that 50% of our streams and rivers are "unsafe" or "dead". It is the purpose of this fact sheet to explain what these facts really mean.

Ohio EPA devotes considerable resources to the monitoring of surface water resources such as streams, rivers, lakes, and wetlands. A systematic framework termed the Five-Year Basin Approach is used. Our goal is to intensively monitor all major watersheds on a rotating cycle of 5-10 years. This includes most of our major rivers, streams, and lakes. By emphasizing biological indicators, such as the fish and invertebrate communities found in streams and rivers, a comprehensive and long term assessment of water resource quality is gained. Aquatic animals are the most consistent and sensitive indicators of environmental quality because they inhabit the water all of the time and respond to all impacts, both chemical and physical. Healthy and flourishing biological communities are also a good indicator that high quality recreational opportunities (*e.g.,* fishing and other outdoor-related activities) are available. By focusing our protection efforts on aquatic life many other important uses (*i.e.,* water supply, recreation) are also covered.

Compared to some States, Ohio uses a comparatively sophisticated and scientifically robust system to determine if rivers and streams meet standards. Because of the variability among States in how this is determined, the information reported nationally by U.S. EPA frequently results in statistics that appear "better" than Ohio's. This problem is discussed in detail in the 1996 Ohio Water Resource Inventory, Executive Summary. Recently, U.S. EPA has followed Ohio EPA's lead by developing new guidelines for States to follow. The goal of this effort is to have more comparable and reliable statistics reported by all States in the future.

Ohio EPA uses biological criteria to rate the quality of our waters . These criteria are used to determine the degree to which standards are exceeded, met, or missed. For communication purposes a rating system has been established. **Exceptional** is the highest quality rating. This rating is given to those sites with the highest species diversity and frequently includes populations of rare, threatened, or endangered species. These waters also support the best sport fisheries. **Good** is assigned to sites with a diversity and quality of aquatic species typical of reference streams and rivers. This varies by ecoregion of which Ohio has five. Streams and rivers that do not meet these standards are considered impaired and are placed into one of two categories, partial attainment and non-attainment. An impaired condition does not mean that the stream or river is "dead" or "unsafe", but rather represents varying degrees of unacceptable condition.



Just as exceptional and good are used to indicate the degree to which Ohio's standards are met or exceeded, three additional ratings are used to indicate the degree to which these standards are missed. **Fair** means that certain characteristics are missing which reflect an "imbalance" in the aquatic community. While many fish, invertebrates, and other forms of aquatic life are generally present, overall diversity is in decline, and species tolerant to nutrients, habitat destruction, and low levels of dissolved oxygen predominate. Placement in the fair category does not mean that the water is unsafe or recreational opportunities do not exist. However, the quality of such opportunities is diminished compared to exceptional and good. **Poor** means that desirable attributes are altogether absent and environmental conditions have worsened. Toxic effects are more prevalent and include declines in species diversity, fewer and smaller fish, fewer invertebrates, and a higher rate of anomalies (lesions, eroded fins, tumors, deformities) on fish. **Very poor** means that environmental conditions have worsened further and that extreme reductions in diversity and abundance have occurred. Other symptoms may include acutely toxic levels of chemicals, complete destruction of habitat, and generally unsafe conditions. Waters rated as **poor** and **very poor** are likely not to support uses important to Ohioans and some of the problems may pose serious health risks.

Nearly one-half (49.3%) of Ohio rivers and streams exhibit **good** or **exceptional** quality. This means that biological communities like those found at background reference sites occur. As such, these waters are also likely to support many other uses important to Ohioans. Partial attainment means that at least one of the biological indicators (fish or invertebrates) exhibits only **fair** quality. In such cases certain sensitive species may be absent or there are too many pollution tolerant species (*e.g.,* carp). Non-attainment means that *all* of the biological indicators are no better than **fair** or one or both groups exhibit **poor** or **very poor** quality. As illustrated by Figure 1, 23.3% of river and stream miles are in partial attainment and 27.7% in non-attainment. When these two categories were separated by major causes of impairment, the partial category (which corresponds to fair quality) was mostly affected by habitat (49.5%) and nutrient enrichment (41.1%), with only a minor fraction (9.4%) caused by toxic pollutants. Non-attainment (which corresponds mostly to poor and very poor quality) was also predominated by habitat (48.6%) and nutrient enrichment (25.8%), but toxic pollution was a larger contributor (25.5%). The presence of toxic pollution as a major cause is an indication that these waters are less safe for other uses than are those affected by habitat and nutrient enrichment. Based on these statistics less than one-tenth (9.2%) of Ohio's rivers and streams are seriously impaired by toxics.

The protection and restoration of aquatic ecosystem quality using Ohio's biological standards has both realized and potential benefits for Ohio. The long-term recreational and economic quality of our waters is strongly linked to that required by aquatic life. For example, the failure to meet these standards due to excessive nutrients and sediment from runoff is frequently correlated with the unacceptable loss of our soil resources via erosion. Habitat destruction such as the clearcutting of trees along streams is evidence that we are attempting to use areas that are more susceptible to flood damage. The degradation of habitat in small, headwater streams makes water quality less suitable for "downstream" uses (*e.g.,* drinking water) and contributes to increased flooding. Thus, a failure to meet aquatic life standards suggests that we may experience "environmental infrastructure" problems in the future.

Human Health Risks

Our assessments include two other indicators that relate directly to human health, fecal bacteria and toxic chemicals in fish tissue. More than one-half (56.9%) of Ohio's rivers and streams are free from bacterial contamination. The remaining waters (43.1%) show levels that indicate varying degrees of risk for human uses such as swimming, canoeing, boating, and wading. The period of greatest risk is usually immediately following rainfall and increased runoff. Bacteria contamination usually reaches streams and rivers via

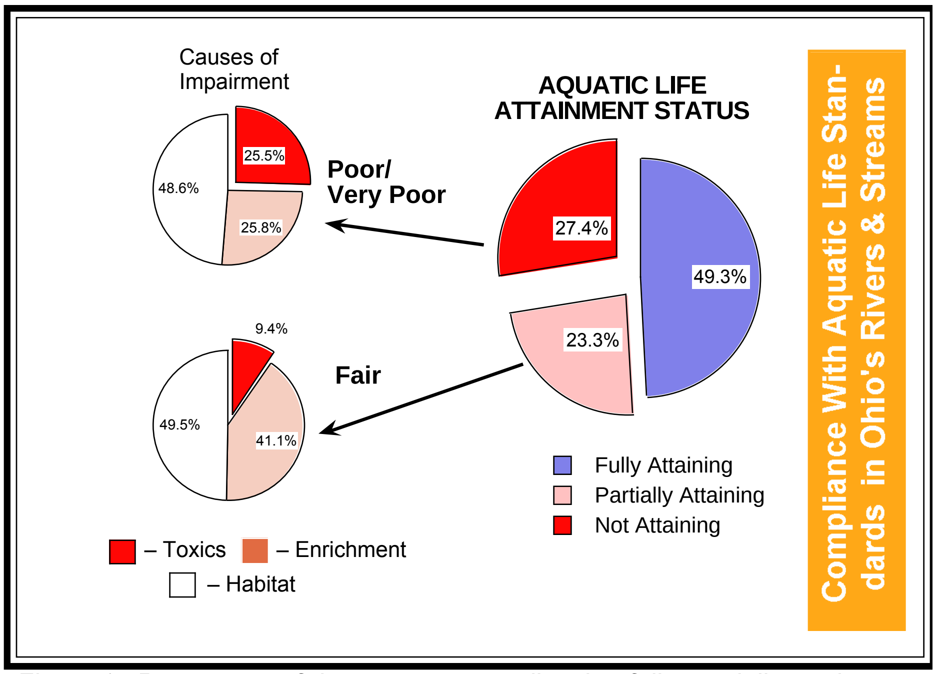


Figure 1. Percentage of river ans stream miles that fully, partially, or do not attain Ohio standards for aquatic life (right) with the major causes of impairment (toxics, habitat, nutrient enrichment) listed for the partial (lower left) and non-attainment categories (upper left).

storm sewers or combined sewer overflows, the effluent of which may contain diluted raw sewage. This problem occurs mainly in the larger urban areas of Ohio, there may be similar problems in unsewered communities. Bacterial contamination can also affect inland lakes and Lake Erie. If public beaches are present, advisories are posted by the Ohio Department of Health when bacterial levels exceed safe thresholds.

Data on the levels of chemicals in fish tissue are used to establish consumption advisories. The monitoring program was expanded in 1993 to include nearly 300 sites sampled each year. In addition, new criteria for consumption advisories recently became available. Four advisory levels establish restrictions on fish consumption as follows: 1) one meal a week, 2) one meal per month, 3) six meals per year, and 4) do not eat. These are based on the levels of certain chemicals (*e.g.,* PCBs, mercury) found in fish with the advisory becoming more restrictive at higher levels. The Ohio Department of Health recently released updated advisories based on the data collected since 1993. Advisories for frequencies of *less* than once per week were listed for 23 Ohio streams, rivers, and lakes. All of the advisories are specific to individual fish species. For example, the consumption of channel catfish may be restricted to one meal per month in a given water, but all other species may have no or lesser restrictions. A statewide advisory for mercury of one meal per week applies to the more sensitive parts of the human population such as women of child-bearing age and young children. Outside of this precautionary statewide advisory for mercury, 18.4% of the stream and river miles monitored had highly or extremely elevated levels of chemicals (one meal per week or six meals per year) for at least one fish species; only 3.8% have a consumption advisory that extends to all species.

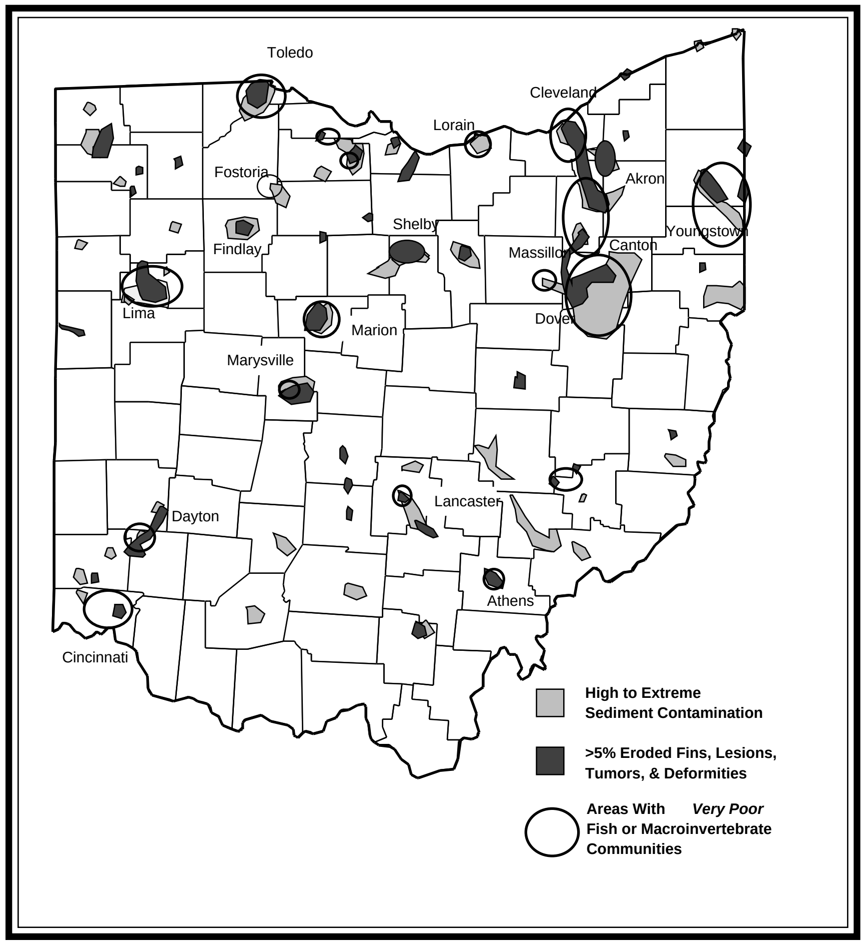


Figure 2. Areas of Ohio with high to extreme sediment contamination, elevated levels of anomalies on fish, and very poor biological communities. The occurence of two or more generally indicates toxic pollution and an increased risk to human health.

Are Ohio's Waters Safe?

Based on the information available to Ohio EPA, the majority of our rivers and streams are safe for activities such as fishing, boating, canoeing, swimming, and wading, even though not all meet standards. However, in a small proportion of rivers and streams, activities such as swimming and eating fish should be restricted to varying degrees. The greatest risks will occur in waters where severe toxic effects are evident (Figure 2). This includes less than 10% of Ohio's river and stream miles. Furthermore, these problems tend to be concentrated in the larger urban areas (Figure 2) and are frequently the result of past activities which occurred prior to recent environmental regulations. Remediating these problems presents a significant challenge. More detailed information about these and other problems is described in the 1996 Ohio Water Resource Inventory, Volume I and other Ohio EPA publications.

For more information contact: Ed Rankin, Division of Surface Water
1800 WaterMark Drive, Columbus, OH 43215-1099
(614)-728-3388;
e-mail: ed.rankin@epa.state.oh.us

Fact Sheet

An Evaluation of Spatial Monitoring & Assessment Design: Preliminary Results from the E. Corn Belt Plains REMAP Project

Ohio EPA employs a targeted, synoptic watershed design for monitoring the chemical, physical, and biological water quality of the State's rivers and streams. This approach is implemented through the Five-Year Basin Approach and is targeted to assess all water quality issues within each targeted watershed or study area. A criticism of this approach is that it may produce biased assessments of the spatial extent of water quality conditions. Furthermore, there is a perception that the five-year basin design inherently targets waters where problems are either suspected or known to exist with a further bias towards point sources of pollution. As such, the aggregated results of the Ohio EPA basin surveys may not truly represent the spatial extent of water quality conditions across Ohio. It is further presumed that the aggregate condition of Ohio's waters are better than that reported in the biennial Ohio Water Resource Inventory (305b report).

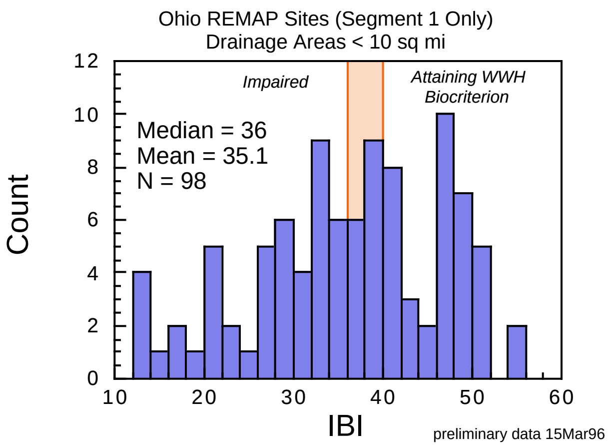
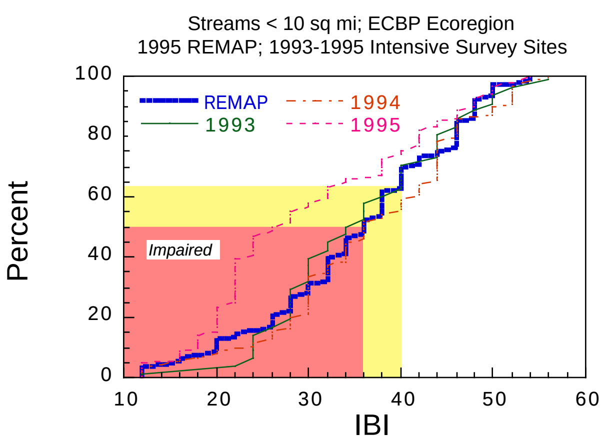


Figure P-1. Frequency histogram of IBI scores from 98 REMAP stations in Ohio with drainage areas < 10 sq mi. Samples collected from June-October 1995.

Recently, U. S. EPA, Region V, and the States of Ohio, Indiana, and Michigan collaborated on a project that was designed to provide a spatially unbiased estimate of aquatic life conditions in the small streams of the Eastern Corn Belt ecoregion within each of these three states (REMAP - Regional Environmental Monitoring and Assessment Program). The estimates of stream quality resulting from this effort are considered to be unbiased because sampling sites were selected using a probabilistic (*i.e.,* random) design. Fish communities (Index of Biotic Integrity, etc.), habitat quality (QHEI), and basic chemical/physical field parameters were collected at each site one time during the period July-early October, 1995.




```
REMAP:   Median - 36; Mean 35.1, N = 98
1993:    Median - 34; Mean 35.3, N = 39
1994:    Median - 36; Mean 36.4, N = 82
1995:    Median - 26; Mean 29.9, N = 75
```

Figure P-2. Cumulative frequency histogram of IBI scores from REMAP stations in the ECBP ecoregion of Ohio during 1995 and intensive survey sites from the same ecoregion in 1993, 1994, and 1995. All sites had drainage areas < 10 sq mi.

The Ohio EPA has intensively sampled the State's rivers and streams since the late 1970s and, as such, has developed an extensive database consisting of more than 5,000 sampling sites. Of the river and stream sizes included in this database, small

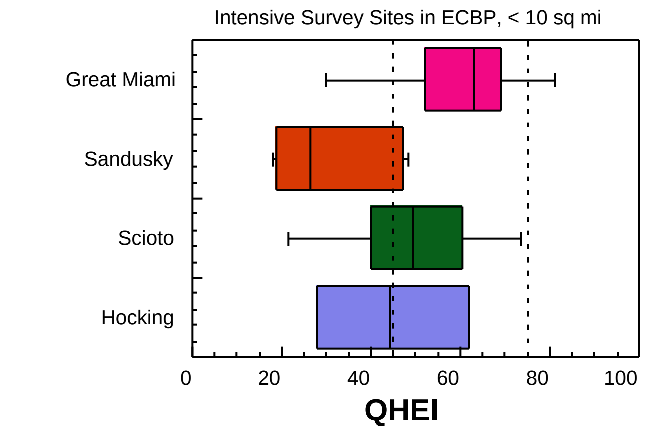


Figure P-2. Cumulative frequency histogram of IBI scores from REMAP stations in the ECBP ecoregion of Ohio during 1995 and intensive survey sites from the same ecoregion in 1993, 1994, and 1995. All sites had drainage areas < 10 sq mi.

streams draining less than 10-20 square miles represent the least sampled in terms of the proportion of stream miles assessed. Approximately 10-15% of the small stream miles have been assessed over this time period compared to more than 50% of streams and rivers draining more than 50 square miles and nearly 100% of those draining more than 1000 square miles. The ECBP REMAP project presented a good opportunity to compare the results of two different spatial sampling designs. This fact sheet is a summary of some preliminary findings for Ohio streams draining less than 10 square miles and to address questions of potential bias in our basin survey data (*i.e.,* how different are synoptic vs. probability estimates of aquatic life condition).

The distribution of IBI scores from the 98 REMAP sites located in Ohio are illustrated in Figure P-1. The median IBI score for these sites was 36 (*i.e.,* the minimum IBI score considered to attain the Warmwater Habitat use designation) which means that 50 percent of the sites are impaired. The distribution also shows a skewness towards lower IBI scores (Figure P-1). This estimate of the proportion of impaired small streams agrees well with the statewide basin survey estimate of stream quality in both the 1994 and 1996 305b assessment cycles for small ECBP ecoregion streams with drainage areas < 10 square miles (50% impaired based on 85.2 miles assessed). Thus the unbiased REMAP design and the spatially biased basin survey designs produced similar estimates of the proportion of impaired small streams in the Ohio portion of the ECBP ecoregion.

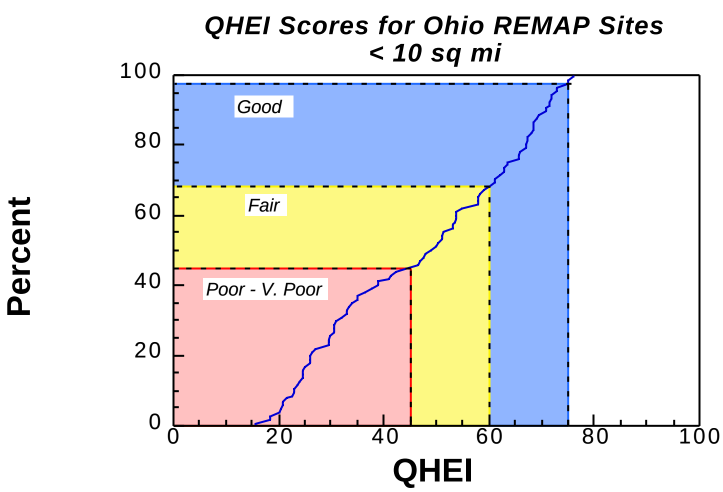


Figure P-4. Cumulative frequency histogram of QHEI scores from REMAP stations in the ECBP ecoregion of Ohio during 1995. All sites had drainage areas < 10 sq mi.

A more direct comparison was made by comparing all IBI scores from the ECBP ecoregion for small streams, by basin survey year, as a cumulative frequency distribution versus the 1995 REMAP results (Figure P-2). The REMAP results were not appreciably different from the basin survey IBI scores in 1993 or 1994, but were different from the 1995 results. The 1995 results were most from the "Clayey, High Lime Till Plains" subregion of the ECBP ecoregion which is characterized by extensive channel modification and impacts from row crop agriculture. The poorer habitat quality of these sites was largely responsible for the much lower median IBI scores in 1995 (Figure P-3). The highest IBI scores were from the comparatively higher quality Twin Creek subbasin. Thus for any given basin year there can be some differences in aggregated use attainment estimates between a randomized and targeted basin survey design. However, when averaged over multiple years the estimates produced by either design were in much closer agreement. Furthermore, these results indicate that the basin survey design employed by Ohio EPA produces an essentially unbiased estimate of small stream quality. The major difference between the REMAP and basin survey design is that the former requires one year to produce a reliable estimate whereas the latter appears to require 4-5 years.

The Ohio EPA basin survey results have increasingly highlighted habitat degradation and sedimentation as major causes of impairment to aquatic life in Ohio's streams. Habitat data (QHEI scores) were collected during the REMAP project, thus an estimate of the extent of habitat degradation can be obtained. Previous work has shown a strong relationship between the condition of fish communities (*i.e.,* IBI scores) and habitat quality in Ohio as measured by the QHEI (Rankin 1995). The 1996 Ohio Water Resource Inventory (305b report) identified habitat degradation and sedimentation as the second and third leading causes of impairment, respectively, statewide. The REMAP data indicated that 45% of the sampling sites had poor or very poor habitat quality and that less than 35% had what is considered to be good quality habitat. This, too, is confirmation of the prevalence of habitat degradation as a major cause of impairment of small streams in Ohio.

The aquatic life in a stream is a sensitive measure of the overall quality of the resource

For more information contact:
Ed Rankin, Division of SurfaceWater
1685 Westbelt Drive
Columbus, OH 43228
(614)-728-3388
e-mail: ed.rankin@epa.state.oh.us

Fact Sheet

Associations Between the Index of Biotic Integrity and Unionized Ammonia in Ohio Rivers and Streams: A Preliminary Analysis

The main purpose of an aquatic life-based chemical criterion is to protect the aquatic life of a stream, river, or lake in accordance with the goal of the designated use. Biocriteria are a direct measure of the aquatic community and as such represent a direct measure of designated aquatic life use attainment status. Having biocriteria provides the Ohio EPA with a unique method to examine whether existing and proposed chemical criteria are over or under-protective of designated aquatic life uses. Previous studies have attempted to evaluate chemical water quality criteria for certain parameters, such as heavy metals, by comparing instream concentrations with different measures of aquatic community health and well-being. However, no study yet has utilized a fully calibrated and standardized system of biological criteria and a statewide chemical water quality and biological database for this purpose.

Many studies have shown the toxic effects of unionized ammonia on aquatic macroinvertebrates and fish. In many instances in Ohio, negative effects to aquatic life have been strongly associated with exceedances of the Ohio EPA water quality criteria for unionized ammonia. Reductions in loadings of ammonia dischrged from point sources has been observed throughout Ohio to be a key in the recovery of previously impaired aquatic life uses. While ammonia was a major cause of impairment in more than 1100 miles of rivers and streams in the 1988 Ohio Water Resource Inventory (305[b] report), this figure had shrunk to 150 miles by 1996.

The purpose of this fact sheet is to examine the association between one of the biological indices which comprises the Ohio EPA biological criteria, the IBI, and unionized ammonia to determine above which ammonia concentrations is aquatic life at risk. A scatter plot of unionixed ammonia based on grab samples collected from Ohio rivers and streams versus the IBI yields a "wedge" of data points (Figure 1). The outer, sloped surface of points approximates the maximum concentrations that have been observed to coincide with a given level of aquatic community performance as portrayed by the IBI. A line drawn on the outer surface of the data points so that 95% of the points fall to the left or beneath the line is referred to as the "95% line of best fit. In the IBI and unionized ammonia example this represents the typically occurring maximum unionized ammonia concentrations at which a corresponding IBI value exists in the statewide database. Chi-square tests of independence were used to test whether or not the occurence of IBI scores are independent of unionized ammonia concentrations at the same sites. If the IBI is independent of the ammonia concentrations, then we can conclude that ambient concentrations of ammonia are not strongly affecting the IBI or the relationship is obscured by other environmental factors. If however, IBI and ammonia are statistically correlated, further analysis to determine the concentrations of unionized ammonia at which a reasonable risk of harm to aquatic life exists should take place.

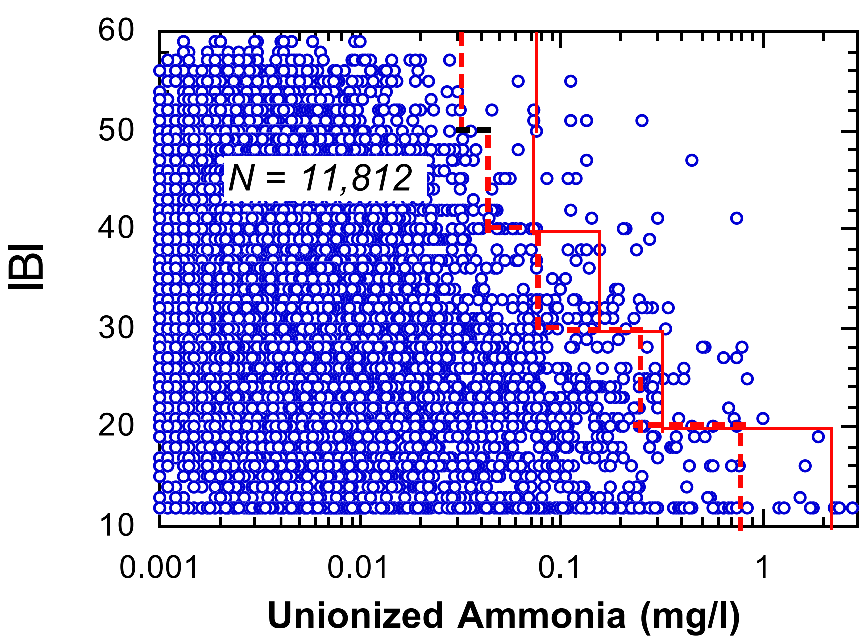


Figure 1. The IBI versus unionized ammonia from streams and rivers monitored by Ohio EPA between 1982 and 1994.

An alternative to generating a "continuous" 95th percentile regression line is to focus more on identifying outliers and extreme values (extreme percentiles) that represent an unacceptable risk to aquatic life. The method to identify outliers and extremes in the data is to cluster the distribution of the independent variable by ranges of IBI scores that correspond to narrative ratings of quality (*e.g.*, exceptional, good, fair, poor, very poor) and the tiered system of aquatic life use deignations employed by Ohio EPA. The upper tenth percentile of the parameter concentration in each IBI category is used to identify the outliers and extremes in each distribution because the biological results at these sites are most likely affected by concentrations of that parameter. Box-and-whisker plots and percentile plots are then used to illustrate the

Table 1. Chi-square test of association between the IBI and un-ionized ammonia based on data collected in Ohio streams between 1982 and 1994 showing actual and expected (in parentheses) observations.

	Un-Ionized Ammonia (mg/l)				
IBI Range	<0.01	0.01-0.05	0.05-0.10	0.10-0.50	> 0.5
50-60	944 (815)	60 (148)	4 (22)	4 (23)	0 (3.3)
40-49	2464 (2203)	251 (401)	12 (61)	9 (63)	1 (9.0)
30-39	2912 (2765)	449 (504)	38 (76)	36 (79)	0 (11.3)
20-29	2377 (2583)	609 (471)	108 (71)	105 (73)	10 (10.6)
12-19	812 (1142)	363 (208)	100 (31)	116 (32)	28 (4.7)

$X^2 = 1135$; $P < 0.0001$

upper, empirically observed values for the independent variable compared to the narrative ranges of the IBI. Outliers in the data are those points that are greater than the upper quartile (UQ: 75th percentile) plus 1.5 times the interquartile range (distance between the 25th and 75th percentiles: UQ - LQ). The other statistic used to describe extreme values is the 99.5th percentile of all the data in an IBI category (illustrated as the 95th percentile of the upper 10 percent of the data in Figure 2). Where such data is strongly skewed the 99.5th percentile can be greater than the "maximum" value where outliers are excluded.

The ranges described above and illustrated in Figure 1 and

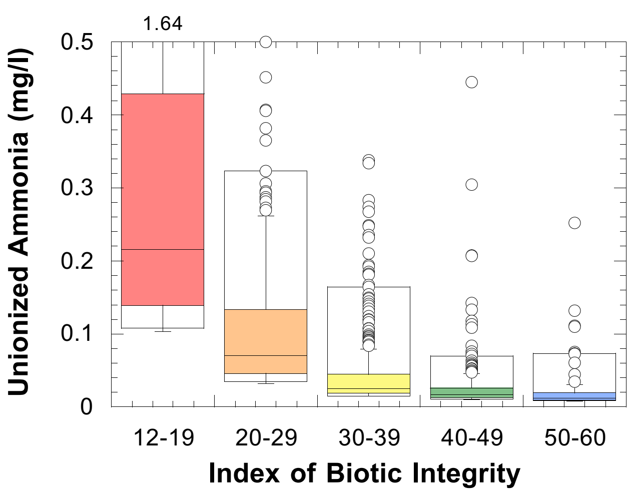


Figure 2. Box-and-whisker and percentile plot of the IBI versus unionized ammonia from streams and rivers monitored by Ohio EPA between 1982 and 1994. Data represent the upper ten percent of the unionized ammonia values within each IBI range. Shaded boxes represent the 25th, median, and 75th percentiles; open boxes the 5th and 95th percentiles; whiskers are the maximum and minimum values excluding outliers which are values greater than the upper (or lower) quartile plus (or minus) 1.5 times the interquartile range.

Table 2 can be used in a risk management approach for establishing water quality criteria, NPDES permit limits, or other water quality managment objectives. Water quality criteria which result in ambient unionized ammonia concentrations in the range of the maximum value, excluding outliers (upper whisker on the plot), and the 99.5th percentile values would be considered to pose an unacceptably high risk to aquatic life and, thus, a lower value should be chosen.

Table 2. Maximum unionized ammonia concentrations (excluding outliers) and 99.5th percentile unionized ammonia values by IBI narrative ranges and corresponding aquatic life uses.

Narrative Range	IBI Range	99.5th %tile Un-ionized Ammonia	Max. Un-ionized Ammonia
Exceptional (EWH)	50-60	0.073	0.031
Good (WWH[1])	40-49	0.070	0.045
Fair (WWH[2])	30-39	0.162	0.080
Poor (MWH)	20-29	0.321	0.262

1 excluding the Huron/Erie Lake Plain (HELP) ecoregion.
2 applies only within the HELP ecoregion.

The scatterplot of unionized ammonia showed a well defined outer boudnary of data points which suggests a strong association with the IBI. The chi-square analysis confirms this association as highly significant (Table1). There were fewer sites that had IBI values >40 (good or WWH) and unionized ammonia concentrations >0.05 than expected (if there were no association) and more sites with low IBI values <30(fair, reflects impairment) and unionized ammonia concentrations >0.05 than expected. The values listed in Table 2 can be used to validate water quality criteria derived by the traditional toxicological approaches. Tiered water quality criteriawhich correspond to the aquatic life uses developed by Ohio EPA have already been established. Other uses of the results presented here could include site-specific applications of the ammonia criteria in combination with the biological criteria. This would be most applicable where instream concentrations exceed the values in Table 2.

Fact Sheet

Associations Between the Index of Biotic Integrity and Total Ammonia-Nitrogen in Ohio Rivers and Streams: A Preliminary Analysis

The main purpose of an aquatic life-based chemical criterion is to protect the aquatic life of a water body in accordance with the goals and objectives of the designated use. Biological criteria are based on measurable attribues of an aquatic community and as such represent a direct measure of designated aquatic life use attainment status. Having biological criteria provides the Ohio EPA with a unique method to examine whether existing and proposed chemical criteria are potentially over or under-protective of designated aquatic life uses. Previous studies have attempted to evaluate chemical water quality criteria for selected parameters, such as heavy metals, by comparing instream concentrations with different measures of aquatic community health and well-being. However, no study yet has utilized a fully calibrated and standardized system of biological criteria and a paired, statewide chemical water quality and biological database for this purpose. This fact sheet describes the observed relationship between a measure of the health and well-being of stream and riverine fish assemblages and total ammonia-nitrogen (N) concentrtions based on data collected between 1982 and 1992 throughout Ohio. This parallels a similar analysis conducted for unionized ammonia-N.

The toxic effects of unionized ammonia-N on aquatic macroinvertebrates and fish are well known. In many instances in Ohio, negative effects to aquatic life have been strongly associated with exceedences of the unionized ammonia-N water quality criterion. Recently, reductions in loadings of total ammonia-N discharged by point sources has been associated with the restoration of previously impaired aquatic life uses in a number of Ohio rivers and streams (Ohio EPA 1997). While ammonia was a major associated cause of impairment in more than 1100 miles (23.9%) of assessed rivers and streams in the 1988 Ohio Water Resource Inventory (305[b] report), this had shrunk to 150 miles (4.5%) by 1996. While the principal deleterious effect of unionized ammonia on fish is toxic, the effect of total ammonia-N on aquatic life reflects both the toxic effects of the unionized fraction of the ammonium ion and the enrichment effect as total ammonia is converted to nitrate. We examined the association between one of the biological indices which comprises the Ohio EPA biological criteria, the Index of Biotic Integrity (IBI), and total ammonia-N to determine whether any relationship is evident. We also examined for the same between total ammonia-N and the number of sensitive fish species. The total ammonia-N data was collected primarily during the summer and early fall months (June - early October), thus the results are mst applicable to this time period. Even though the influence of winter ammonia-N levels is implicity addressed by the biological assessment data, safe levels of winter ammonia-N cannot be derived from this database.

A scatter plot of total ammonia-N based on grab samples collected from Ohio rivers and streams versus the IBI yields a "wedge" of data points (Figure 1) similar in shape to that previously observed for unionized ammona-N. The outer sloped surface of points approximates the maximum concentrations that have been observed to coincide with a given level of aquatic community performance as portrayed by the IBI. A line drawn on the outer surface of the data points so that 95% of the points fall to the left or beneath the line is referred to as the "95% line of best fit". In the IBI vs. total ammonia-N example this represents the typically occurring maximum ammonia concentrations at which a corresponding IBI value exists in the statewide database. Chi-square tests of independence were used to test whether or not the occurence of IBI scores are independent of total ammonia-N concentrations at the same sites. If the IBI is independent of the total ammonia-N concentrations, then we can conclude that ambient concentrations of ammonia are not significantly affecting the

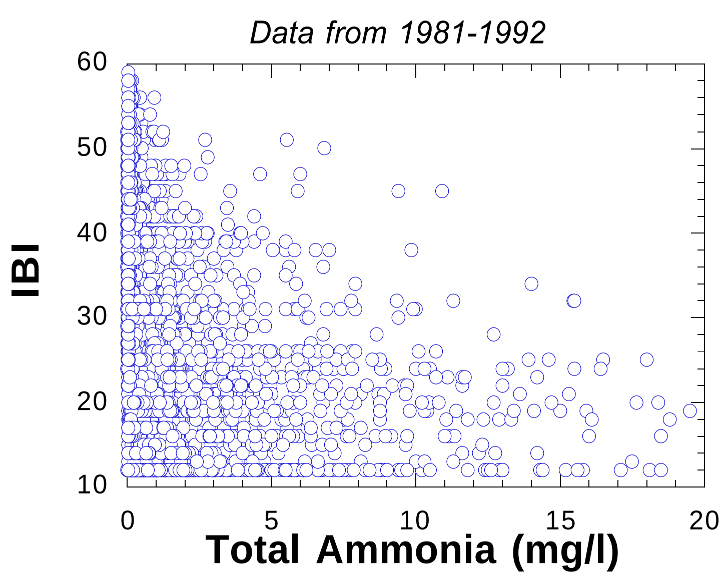


Figure 1. The IBI versus total ammonia-N from streams and rivers monitored by Ohio EPA between 1982 and 1992.

Table 1. Chi-square test of association between the IBI and total ammonia based on data collected in Ohio streams between 1982 and 1992 showing actual and expected (in parentheses) observations.

	Total Ammonia (mg/l)					
IBI Range	≤ 1.0	1.1-2.0	2.1-5.0	5.0-10.0	10.0-15.0	> 15.0
50-60	790 (705)	230 (89)	1 (34.6)	2 (13.3)	0 (4.1)	0 (3.0)
40-49	2671 (2416)	207 (172)	19 (119)	3 (45.6)	1 (14.1)	0 (10.1)
30-39	3118 (3933)	103 (156)	65 (144)	27 (55.4)	3 (17.1)	2 (12.3)
20-29	3162 (3233)	609 (471)	187 (159)	71 (61)	22 (18.9)	9 (13.6)
12-19	1217 (1671)	363 (208)	267 (82)	104 (31.6)	38 (9.8)	35 (7.0)

X^2 = ????; P < 0.0001

IBI or the relationship is obscured by other environmental factors (*e.g.*, proportion of total ammonia that is unionized). If, however, IBI and total ammonia-N are statistically correlated, further analysis to determine the concentrations of total ammonia-N at which a reasonable risk of harm to aquatic life can exist should take place. This does not necessarily indicate direct causality, but rather these relationships can be used to estimate "concentrations of concern".

An alternative to generating a continuous 95th percentile regression line is to focus more on identifying outliers and extreme values (extreme percentiles) that clearly represent an unacceptable risk to aquatic life. The method to identify outliers and extremes in the data is to cluster the distribution of the independent variable (total ammonia-N) by ranges of IBI scores that correspond to narrative ratings of quality (*e.g.*, exceptional, good, fair, poor, very poor) and the tiered system of aquatic life use designations currently employed by Ohio EPA. The upper tenth percentile of the parameter concentration in each IBI range was used to identify the outliers and extremes in each distribution. Box-and-whisker plots and percentile plots were used to illustrate the upper, empirically observed values for the independent variable compared to the five narrative ranges of the IBI. Outliers in the data are statistically defined as those points that are greater than the upper quartile (UQ: 75th percentile) plus 1.5 times the interquartile range (distance between the 25th and 75th percentiles: UQ - LQ). The other statistic used to describe extreme values is the 99.5th percentile of all the data in an IBI category (illustrated as the 95th percentile of the upper 10 percent of the points in Figure 2). Where such data is strongly skewed, the 99.5th percentile can be greater than the "maximum" value where outliers are excluded.

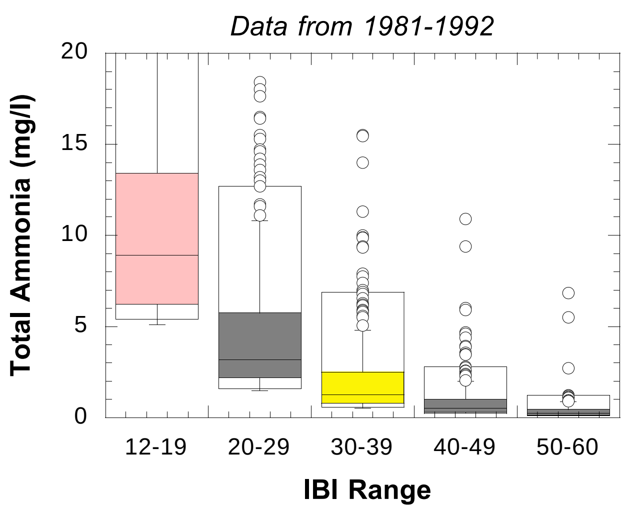


Figure 2. Box-and-whisker and percentile plot of the IBI versus total ammonia (mg/l) from streams and rivers monitored by Ohio EPA between 1982 and 1992. Data represent the upper ten percent of the unionized ammonia values within each IBI range. Shaded boxes represent the 25th, median, and 75th percentiles; open boxes the 5th and 95th percentiles; whiskers are the maximum and minimum values excluding outliers which are values greater than the upper (or lower) quartile plus (or minus) 1.5 times the interquartile range.

Table 2. Maximum total ammonia (mg/l) concentrations (excluding outliers) and 99.5th percentile unionized ammonia values by IBI narrative ranges and corresponding aquatic life uses.

Narrative Range	IBI Range	99.5th %tile Total Ammonia	Max. Total Ammonia
Exceptional (EWH)	50-60	1.24	0.87
Good (WWH[1])	40-49	2.8	2.04
Fair (WWH[2])	30-39	6.9	4.8
Poor (MWH)	20-29	12.7	10.8

1 excluding the Huron/Erie Lake Plain (HELP) ecoregion.
2 applies only within the HELP ecoregion.

The ranges described above and illustrated in Figure 1 and Table 2 can be used in a risk management approach for establishing and modifying water quality criteria, NPDES permit limits, and for other water quality managment objectives. Water quality criteria which result in ambient total ammonia-N concentrations in the range of the maximum value, excluding outliers (upper whisker of the box-and whisker plots), and the 99.5th percentile values would be considered to pose an unacceptable risk to aquatic life.

The scatterplot of total ammonia-N showed a well defined outer boundary of data points which suggests a strong association with the IBI. The chi-square analysis confirmed this association as being highly significant (Table 1). There were fewer sites that had IBI values >40 (good - meets the WWH use) and total ammonia-N concentrations >1.0 mg/l, than what would have been expected if there were no significant relationship. Conversely, there were more sites with IBI values <30 (poor - reflects impairmentof the WWH use) and total ammonia-N concentrations >1.0 mg/l than what would have been expected if there were no significant relationship. The values listed in Table 2 can be used to ground truth water quality criteria derived by the more traditional toxicological approaches. Tiered water quality criteria, which correspond to the aquatic life uses developed by Ohio EPA, have already been established for aamonia-N. Another use of the results presented here would include validating site-specific applications or modifications of the ammonia-N criteria.

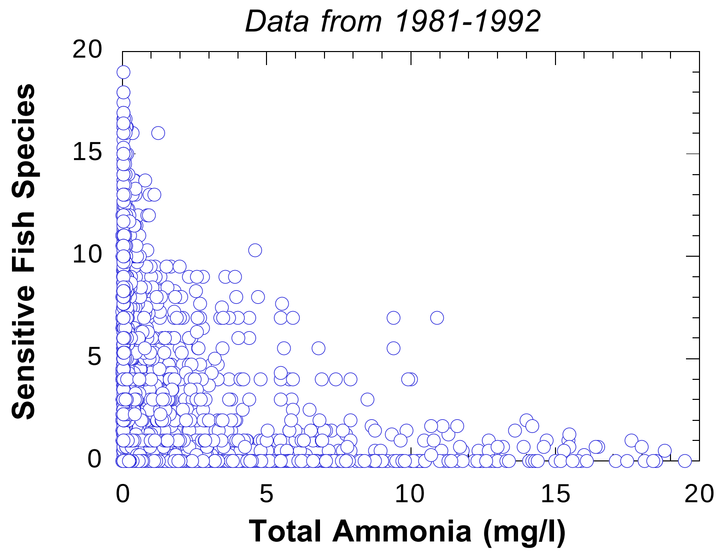


Figure 3. The number of sensitive fish species versus total ammonia from streams and rivers monitored by Ohio EPA between 1982 and 1992.

Figure 3 illustrates the relationship between the number of sensitive fish species and total ammonia-N. The distinct decline in the number of senstive species along an increasing continuum of total ammonia-N (especially >10 sensitive species) reflects the level of sensitivity of the Ohio's highest quality waters. The fish species in these streams and rivers are sensitive not only to toxic effects of ammonia, but also to more subtle shifts in the trophic dynamics of these ecosystems caused by increasing nutrient enrichment. Rivers and streams with more than 10 sensitive fish species usually have total ammonia-N concentrations less than 1.0 mg/l.

Demonstrating Linkages Between Indicators: Scioto River Case Study

ADMINISTRATIVE INDICATORS

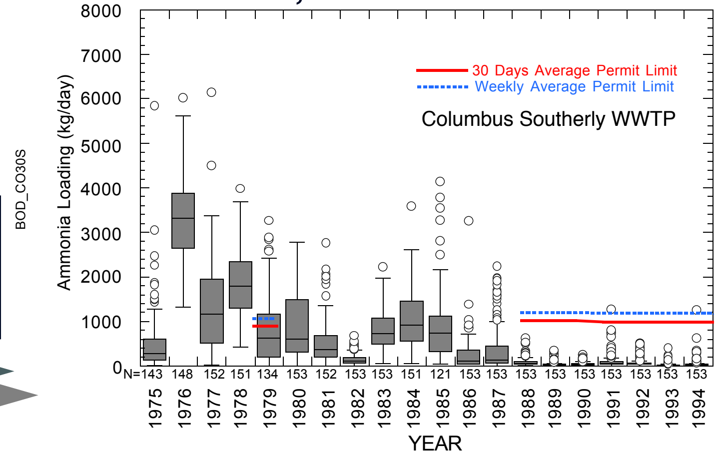
LEVEL 1:
Ohio EPA issues WQ based permits & awards funds for Columbus WWTPs

$$$$
NPDES

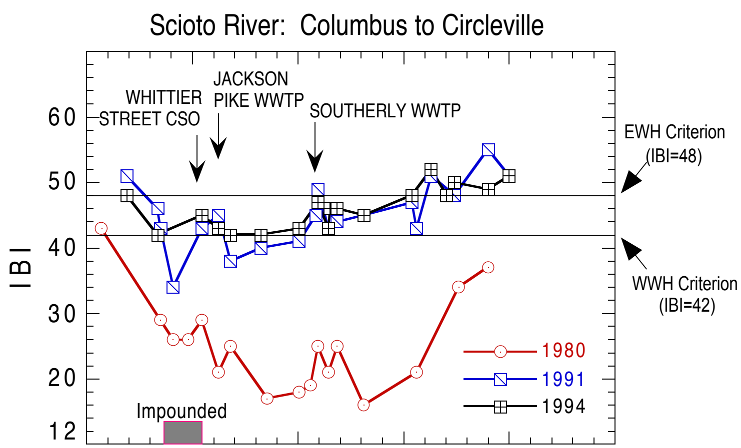
LEVEL 2:
Columbus constructs AWT by July 1, 1988; permit conditions attained

STRESSORS

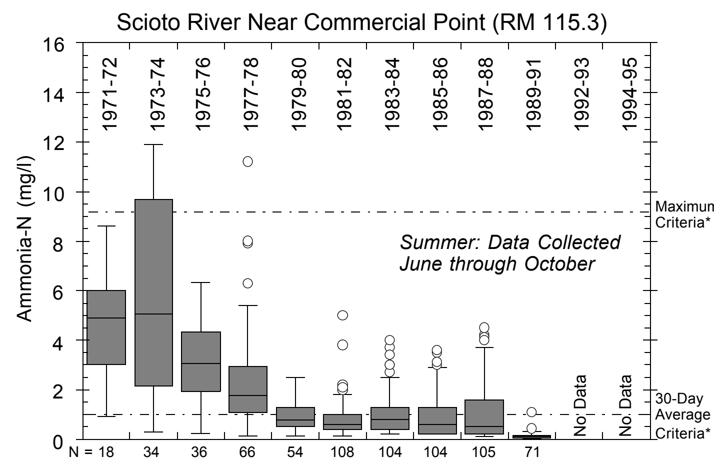
LEVEL 3: Loadings of ammonia, BOD, etc. are reduced



LEVEL 6: Biological recovery evidenced in biocriteria; 3 yrs. post AWT



LEVELS 4&5: Reduced instream pollutant levels; enhanced assimilation



RESPONSE

EXPOSURE

Demonstrating Linkages Between Indicators: Ottawa River Case Study

ADMINISTRATIVE INDICATORS

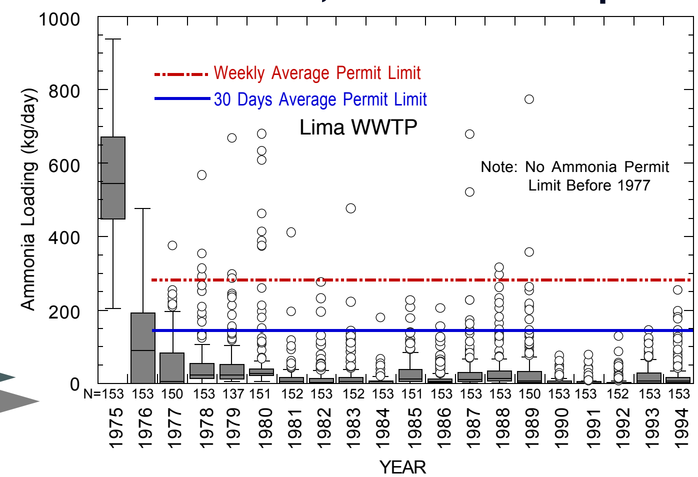
LEVEL 1:
Ohio EPA issues WQ based permits & awards funds for the Lima WWTP

$$$$
NPDES

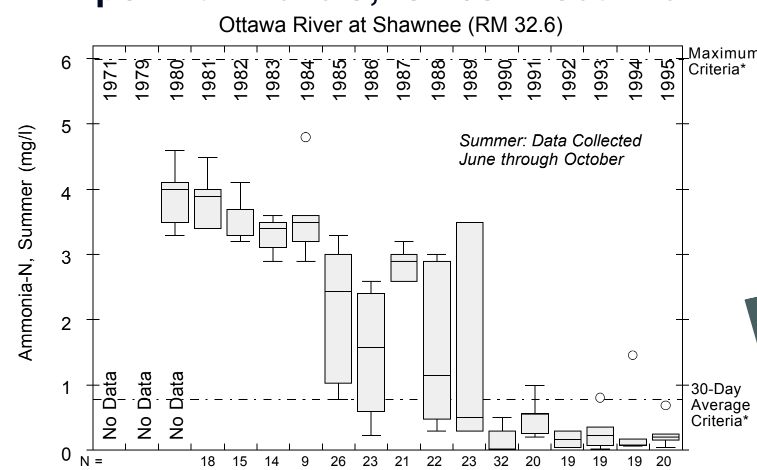
LEVEL 2:
Lima constructs AWT by mid 1980s; permit conditions attained by 1990

STRESSORS

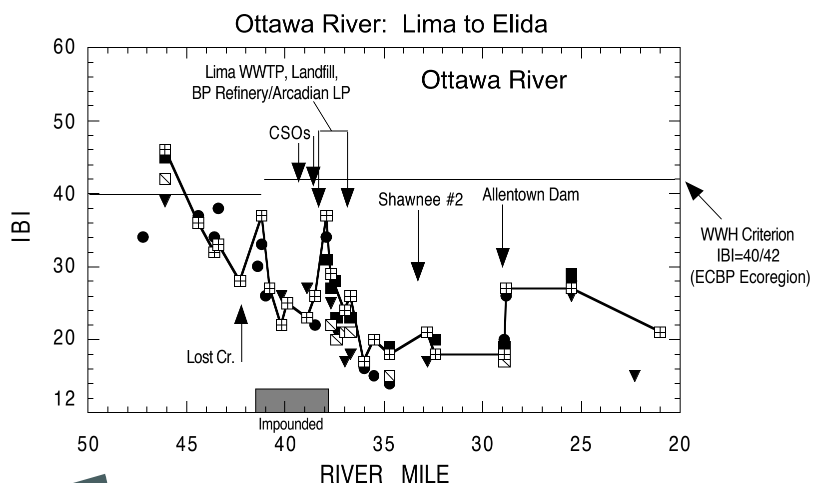
LEVEL 3: Loadings of ammonia, BOD, were reduced; other sources present



LEVELS 4&5: Reduced instream pollutant levels; toxics in sediment



LEVEL 6: Biological recovery incomplete 6-8 yrs. post AWT; toxic response signatures



RESPONSE

EXPOSURE

SANDY CREEK

The water and ecological quality of the upper Sandy Creek has been monitored and evaluated by the Ohio EPA during 1993, 1996, and 1997. The 1996 study included sampling of water quality, sediment quality, fish and aquatic insect communities, stream habitat quality, and fish tissue for contaminants. Three sites downstream from Minerva were resampled in 1997.

STREAM HABITAT

The upper Sandy Creek is comprised of a good to excellent mixture of pool, riffle, and run habitats beneficial to supporting good to exceptional biological communities.

FISH CONTAMINATION

Polychlorinated biphenyls (PCBs) were detected in a number of fish, with the highest values reported in common carp. Three carp samples collected upstream from Malvern had PCB concentrations exceeding Ohio water quality standards.

WATER QUALITY

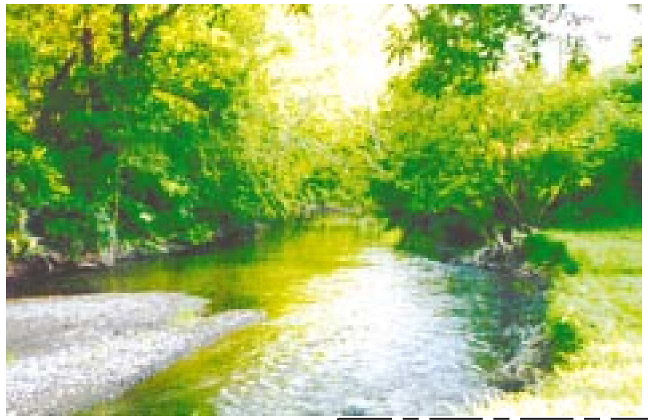
Chemical contaminants caused severely toxic conditions in Sandy Creek during 1996. Potential sources of chemicals include ammonia from the Minerva wastewater plant and unknown compounds spilled or released into the stream. Ammonia from the Minerva wastewater plant was significantly reduced in 1997.

BIOLOGICAL TRENDS

In 1993, severe biological degradation occurred in Sandy Creek immediately downstream from the Minerva WWTP. Biological communities were severely degraded during 1996, but substantial improvement occurred in 1997.

Quick Facts - Sandy Creek

length:	41 miles
gradient:	10.0 feet / mile
river miles assessed:	12.3 miles
fish species:	36
aquatic insect species:	128
Ohio endangered species:	none
aquatic life use designation:	warmwater habitat
average river flow:	175,000,000 gallons/day
fish consumption advisories:	none

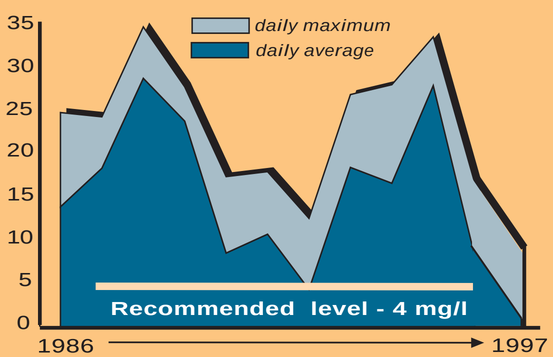


STREAM LIFE QUALITY

- EXCEPTIONAL
- GOOD
- FAIR
- POOR/VERY POOR
- NOT EVALUATED

AMMONIA
A POLLUTANT

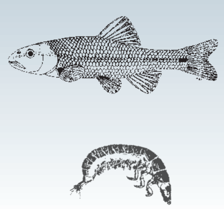
Ammonia discharged into Sandy Creek from the Minerva wastewater plant.



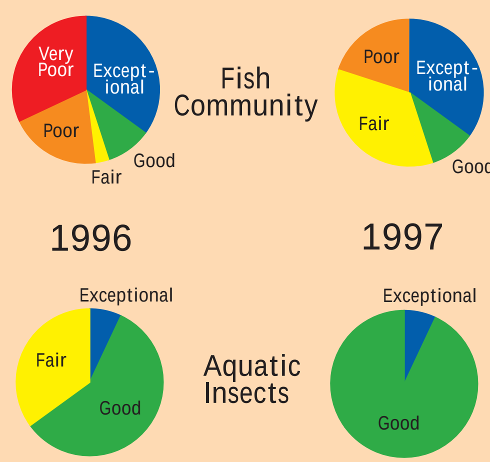
Pollution Sensitive Species

Some of the more common aquatic species in Sandy Creek which are indicative of clean water and good habitat.

- river chub
- hornyhead chub
- rosyface shiner
- banded darter
- mayflies
- caddisflies

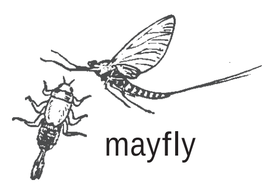


STREAM HEALTH

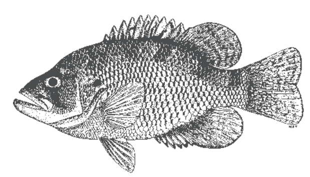


Clean Water Act Goals

	1997	1996
stream miles fully meeting:	6.9	5.4
stream miles partially meeting:	5.0	0.3
stream miles not meeting:	0.4	6.6



mayfly



rockbass

Ohio EPA

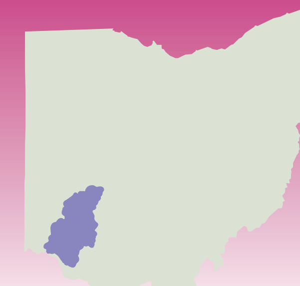
Ohio Watersheds

The Little Miami

State and National Scenic River

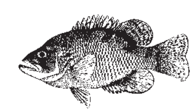
A guide to Ohio's Streams

The Little Miami watershed occupies 1,757 square miles of ten southwestern Ohio counties. Originating near South Charleston, the mainstem flows in a southwesterly direction to its confluence with the Ohio River near Cincinnati. The watershed contains 133 named streams, some of Ohio's most scenic and diverse riverine habitats, and a high diversity of aquatic organisms (including six Ohio endangered species). With canoe liveries, bike trails, parks, and healthy populations of sport fish at many locations, it is easy to understand why the Little Miami River is a popular recreational retreat for Ohioans.



Clean Water Act goals

stream miles meeting: 41
stream miles partially meeting: 58
stream miles not meeting: 3



Fish Species: 87

water quality

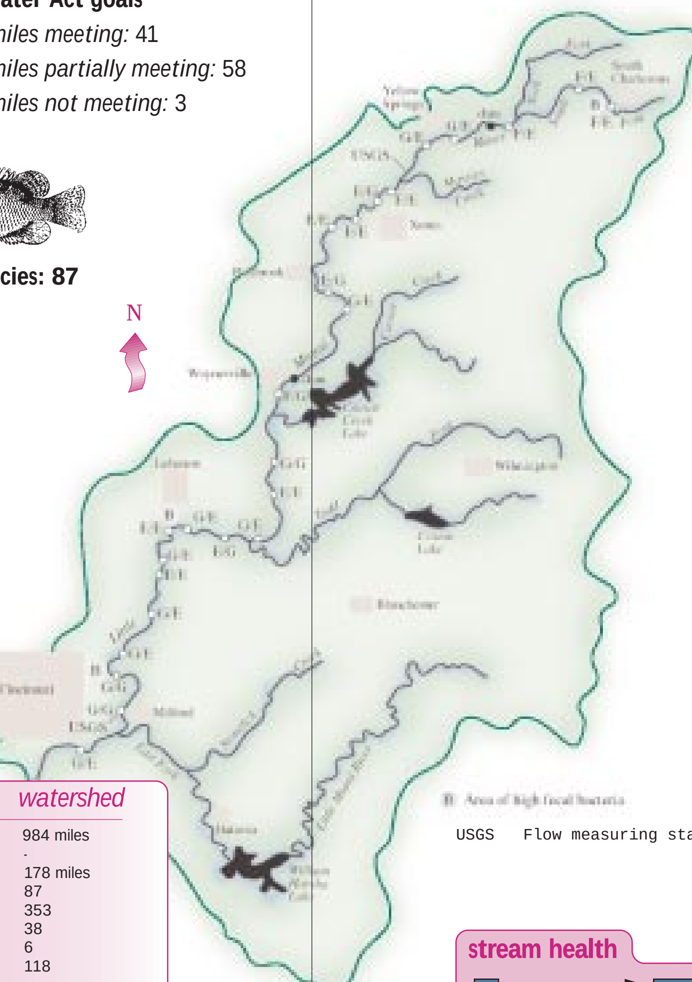
major problems	*major sources*
organic/nutrient enrichment	municipal sewage
fish abnormalities	suburbanization
siltation	row crop agriculture, new construction, eroding streambanks
river bacteria	municipal sewage, sewer overflows to the river, urban runoff, livestock

Water quality trends

The Little Miami River has shown a significant improvement in water quality since the 1980's. Sewage containing organic material has been substantially reduced from most wastewater plants - a direct result of improved treatment. However, the total amount of pollutants still exceeds the capacity of the Little Miami River to adequately assimilate the waste.

quick facts

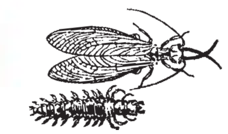
	mainstem	*watershed*
length:	105.5 mile	984 miles
gradient:	6.5 feet / mile	-
river miles assessed:	102 miles	178 miles
fish species:	83	87
aquatic macroinvertebrate taxa:	268	353
(mussel species):	36	38
Ohio endangered species:	6	6
scenic river miles:	105	118
aquatic life use designation:	EWH	-
wastewater dischargers:	5	24
wastewater volume:	12 MGD	50 MGD
average river flow:	811 MGD	-
public access sites:	13	-
canoe liveries:	7	-
number of dams:	2	3 major reservoirs
fish consumption advisories:	none	none



USGS Flow measuring station

Recreational Opportunities

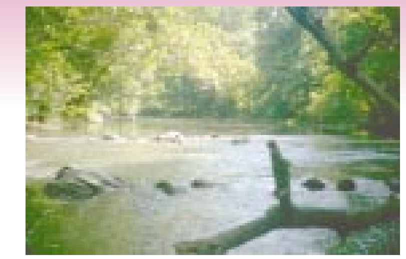
sport fishing: smallmouth bass, rockbass flathead catfish, sauger
canoeing: 250,000 people per year
biking & hiking trails: 50+ miles
parks & wildlife areas & preserves: 12?



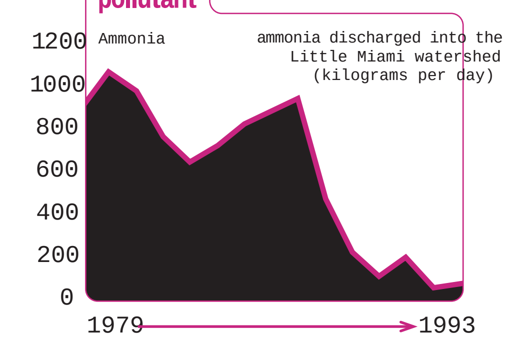
Macroinvertebrate Taxa: 353

Stream Habitat

The Little Miami River is comprised of a good mixture of pool, riffle and run habitats. The river channel has been little modified by man, with less than four percent affected by dam impoundments or channelization. The Little Miami River and its tributaries contain some of the highest quality river habitat in Ohio.



pollutant

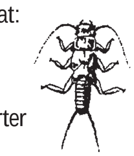


Special highlights

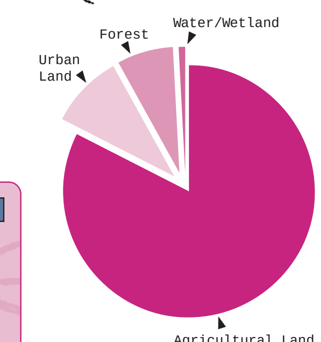
Ohio's first state and national designated scenic river Ohio's longest exceptional warmwater habitat stream A reproducing population of blue suckers, one of Ohio's rarest and most endangered fish species, were collected for the first time in the Little Miami River during 1993

pollution sensitive species

Some of the more common aquatic species in the Little Miami River which are indicative of clean water conditions and good habitat:



black redhorse
slenderhead darter
stoneflies
shorthead redhorse
rainbow mussel
mayflies

stream health

	fish community	aquatic insects
exceptional	42 miles	85 miles
good	37 miles	14 miles
fair	23 miles	3 miles
poor	0 miles	0 miles
very poor	0 miles	0 miles

timeline

1846 Little Miami railroad completed

1913 Major flooding occurs during March

1942 Major fish kills in Little Miami River. Health Commissioner declares fish are not safe to eat

1967 Little Miami, Inc., a nonprofit citizen organization, is formed to preserve the Little Miami River

1969 Upper Little Miami River designated Ohio's first State Scenic River by ODNR

1971 Entire Little Miami River becomes State Scenic River with designation of lower section by ODNR

1973 Upper Little Miami River designated first Federal Scenic River. First Little Miami River land protection site acquired by ODNR, Scenic Rivers program

1974 Little Miami Railroad is abandoned

1979 ODNR acquires abandoned Little Miami Railroad

1980 Lower Little Miami River designated a Federal Scenic River

1983 Ohio EPA study shows less than 2 miles of Little Miami River meet Clean Water Act goals

1986 Ohio endangered Mountain Madtom fish collected in Little Miami River. ODNR removes lowhead dam at Foster

1988 Major improvements made at many wastewater treatment plants in Little Miami River watershed

1993 Ohio EPA study shows over 40 miles of Little Miami River meet Clean Water Act goals. Ohio endangered blue sucker fish collected in Little Miami River near Cincinnati