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Comparison of Biological-based and Water Chemistry-based Aquatic Life Attainment/Impairment Measures under a Tiered Aquatic Life Use System

Aquatic Life Use Attainment

Fact Sheet 3-CABB-03

Aknowledgements

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Assessing Aquatic Life Use Attainment

Aquatic Life Use Attainment at Stream Stations in Ohio: Biocriteria vs. Chemical Criteria Based Methods

Impacts on Listing Waters for 305(b) and TMDL Lists

Background

The Clean Water Act (CWA) has an ultimate goal of achieving biological integrity. Engineering solutions to point source impairments have largely succeeded in reducing point sources of impairment. A basis for this success has been the derivation of water quality criteria, adopted by each State to protect the various designated uses (e.g., aquatic life, recreation) that act as the design goals for treatment processes or for best management practices (BMPs). Much of the early tracking of success, besides issuance of NPDES permits, was focused on the water chemistry criteria used as the design goals of the permitting process. As point sources impairments have diminished, biological monitoring has been increasingly used as an important way to assess and understand the complex affects of multiple stressors including non-chemical stressors. A strength of bioassessment and associated biocriteria is that they can integrate the effects of all stresssors on biological integrity, a major goal of the CWA.

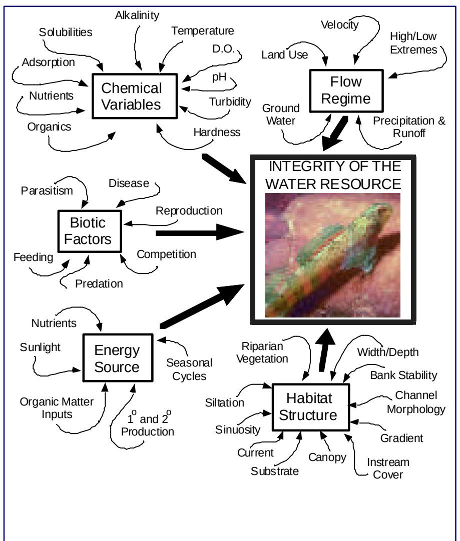


Figure 1. Conceptual model of the five major factors that affect water resource integrity in streams. Modified from Karr et al. 1986.

Water chemistry data is still an essential component of any monitoring program. Because many States still rely on water chemistry data to assess aquatic life use attainment for 305(b) reporting and 303(d) listing, it would be useful to know the strengths and limitations of water chemistry-based vs. biocriteria-based assessment processes. Ohio EPA has used an intensive survey approach integrating biological data and water chemistry along with other data in assessing condition and causes of impairment and threats to their streams and rivers for the past 20 years. Much of this data is in electronic databases that provides an opportunity to compare and contrast attainment decisions where both data types co-occur. The purpose of this fact sheet is to compare, in a retrospective fashion, aquatic use attainment using a biocriteria-based approach with a water chemical criteria-based approach as outlined in US EPA's 305(b) guidance document (U.S. EPA 1997a,b).

Ohio Data

The biological data used in these analyses were collected by Ohio EPA staff or other acceptable scientists that followed the Ohio EPA sampling and taxonomic protocols (Ohio EPA 1987b, DeShon 1995). The period of data collection spans the late 1970s to 2000 and includes biological assemblage data collected on fish and macro-invertebrates using standardized methods (Ohio EPA 1987b), grab samples of conventional and toxic chemicals analyzed with approved U.S. EPA methods, and habitat data based on the Qualitative Habitat Evaluation Index (Rankin 1989, 1995, Ohio EPA 1989b). Some of the watersheds in these regions have been sampled two or three times over this time span. Water chemistry data has been collected at many of these sites over the same time span and habitat data is available from the mid 1980s to the present. The water quality criteria for this study are the current Ohio Water Quality Criteria and the guidelines for making use attainment decisions with this data are based on the U.S. EPA guidance for 305(b) reports (USEPA 1997a). For nutrient parameters that do not yet have aquatic life criteria, we used background reference values to identify sites with highly elevated total phosphorus levels for analyses of causes or stressors among the various scenarios we discuss. These values were based on analyses of associations between nutrients, biological assemblages, and habitat (Rankin et al. 1999). Sites sampled to characterize effluents or mixing zone sites were excluded from analyses. Data is stored in a FoxPRO format and Ohio EPA programs were used or modified to calculate chemical and biological attainment status by sampling station.

A station identifier was created for each common study site in the Ohio database. Fish and/or macroinvertebrate sampling sites at the same or nearby sites were linked along with habitat (QHEI) data and water chemistry data sampled during the same summer period in the same year. Linking was done on a case-by-case basis to ensure that the effects of dischargers and other pollution sources, habitat, and confluences were similar among data types. This data was also linked to Ohio's Waterbody System (305b) database by sampling reach. Sites were assigned to the matching waterbody river reach and linked to the major causes of impairment identified for the reach as part of Ohio's watershed and intensive survey program. These assessments summarized the results of the databases being used above as well as addition data on pollution sources, sediment contamination and toxicity, permit information, fish kills, chemical spills, and BPJ of biologists, scientists, and engineers at Ohio EPA. Assignment of causes of impairment follow Ohio EPA protocols for 305(b) assessments (Ohio EPA 1996, 1998) which closely follow guidelines of U.S. EPA (1997a). Identification of impaired waters for 305(b) reporting in Ohio largely followed a weight of evidence approach.

Ohio's Biocriteria

Ohio has pioneered the use of numerical biocriteria to judge the attainment or impairment of CWA goals (Ohio EPA 1987a,b; 1989a, b). Numerical biological criteria in Ohio are based on multimetric biological indices including the Index of Biotic Integrity (IBI) and modified Index of Well-Being (MIwb), indices measuring the response of the fish community, and the Invertebrate Community Index (ICI), which measures the response of the macroinvertebrate community. The IBI and ICI are multimetric indices patterned after an original IBI described by Karr (1981) and Fausch et al. (1984). The ICI was developed by Ohio EPA (1987b) and further described by DeShon (1995). The MIwb is a measure of fish community abundance and diversity using numbers and weight information and is a modification of the original Index of Well-Being originally applied to fish community information from the Wabash River (Gammon 1976; Gammon et al. 1981). Performance expectations for the principal aquatic life uses in the Ohio WQS (Warmwater Habitat [WWH], Exceptional Warmwater Habitat [EWH], and Modified Warmwater Habitat [MWH]) were developed using the regional reference site approach (Hughes et al. 1986; Omernik 1987; Hughes and Larsen 1988; Larsen et al. 1988; Gallant et al. 1989). This fits the practical definition of biological integrity as the biological performance of the natural habitats within a region (Karr and Dudley 1981). Numerical endpoints are stratified by ecoregion, use designation, and stream or river size. These biological criteria codified in the Ohio Water Quality Standards (WQS; Ohio Administrative Code [OAC] 3745-1-07, Table 7-14). Three attainment status results are possible at each sampling location - Full, partial, or non-attainment. Full attainment means that all of the applicable indices meet the Ohio WQS biocriteria. Partial attainment means that one or more of the applicable indices fails to meet the biocriteria. Non-attainment means that none of the applicable indices meet the biocriteria or, for WWH and EWH streams, one of the organism groups reflects poor or very poor performance.

Biocriteria-based vs. Water Chemistry-based Measures of Aquatic Life Use Attainment

It is an important axiom that the more directly one can measure a use designation the more accurate the assessment typically is. This point was made by the National Academy of Science committee charged with assessing the USEPA TMDL program (National Research Council 2001). Many states, however, still rely heavily on surrogate chemical measures to assess attainment of aquatic life uses. Yoder (1997) laid out the proper roll of various indicators in an adequate monitoring program, however, the errors in using indicators in a potentially inaccurate manner is poorly known. Because of this it is important to examine the differences and error rates of indirect or surrogate approaches compared to more direct measures of aquatic life use attainment (i.e., based on biocriteria). We are using data collected in Ohio over a 20 year period that matches biological data based on fish and/or macroinverbrates with a suite of water chemistry stressors typically used by states to make aquatic life use attainment decisions.

Decisions for determining impairment/attainment of aquatic life uses with biocriteria in Ohio, based in fish and macroinvertebrates, are summarized above and discussed in more detail in Yoder (1995), Yoder and Rankin (1995a,b), Ohio EPA (1987b), and DeShon (1995). Attainment decisions on the basis of water chemistry criteria followed that of the U.S. EPA 305(b) guidance document from 1997 (U.S. EPA 1997a,b). For conventional pollutants (dissolved oxygen, pH, and temperature) we used the 305(b) cutoff of < 10% exceedences (Full Support), 11-25% exceedences (Partial Support) and > 25% exceedences (Non-Support). For toxic pollutants (e.g., metals and ammonia) an exceedence within a three year period is considered non-attainment of the aquatic life use. Since most of the data is collected during only one summer during most Ohio intensive surveys, we felt this was a reasonable approach (as opposed to allow one exceedence in three years to be attaining the use). This fact sheet does not address concerns with data extrapolation or sampling coverage since it is simply comparing biological and water chemistry data collected at the same sampling station within the same year.

Results and Discussion

Trends in Assessment Results

Ohio EPA aquatic life results have been summarized by reach as part of their 305(b) reports from 1988 to 2000. Aquatic life in Ohio streams reaches has improved dramatically over this period which is related to decreasing water chemistry loads from municipal and industrial sources (Ohio EPA 1996, 1998, 2000). This strong trend is evident in the Ohio data set when examined site-by-site as well (Figure 2). Much of this progress has been attributed to progress from abatement of point source pollutants (Ohio EPA 2000). The relation to load reductions is supported by Figure 3 which illustrates a sharp continual decline over the past 20 years in exceedences of important chemicals often associated with point source discharges, including ammonia, copper, cadmium, and dissolved oxygen.

Figure 2 and 3 illustrate that both chemical and biological indicators captured the decreasing trends in point source influence on Ohio streams and rivers. It is important, however, to tease apart the differences in conclusions that might occur from an approach based on water chemistry data alone to one that is built around biocriteria as an attainment/impairment indicator and that uses water chemistry and other stressor data to identify causes of impairment.

Lack of water chemistry exceedences, as a measure of "attainment" of aquatic life uses, a currently acceptable indicator for 305(b) assessments (U.S. EPA 1997). When absence of all chemical exceedences is plotted over time (Figure 4, top panel, orange diamonds) as a measure of aquatic life use attainment it clearly overestimates overall attainment when compared to aquatic life use attainment based on biocriteria (Figure 4, top panel, blue dots). The slopes of these lines are similar, matching the indications that the progress observed in the biological responses was a result of point source abatement. The amount of overestimation is about 30% for any given year. This, in essence, is the error rate that would have occurred if Ohio relied on a simple water chemistry indicator approach.

To provide some more evidence related to causes of impairment over time, we plotted the percent of stations over this time period that had poor or very poor habitat conditions are measured by QHEI scores (Figure 4, bottom panel). There is some variability in this data, but also no clear trend over time. Variation in these QHEI scores is more likely a function of study areas and the tiers of aquatic life uses we were examining in any given year. This cannot be directly translated in a percent of “impairment” related to habitat because the tiered aquatic life uses control for this where a use attainability analysis has shown that a “higher” use in unattainable at the present time. In addition, higher QHEI scores can still result in impairment in high quality (EWH and some WWH streams). The purpose of this figure is to illustrate that progress to restore stream habitat has been minimal compared to water chemistry improvements and that the trends observed in biological attainment are not related to habitat restoration.[1]

We next compared water chemistry vs. biocriteria-based station-by-station measures of attainment of aquatic life attainment and impairment. Figure 5 contains pie charts showing the agreement between biological-based and water-chemistry-based assessment of aquatic life use attainment. Because of the clear trends in attainment over time we broke the analyses into three time periods. The trend among these three time period (before 1988, 1988-1993, and 1994-2000 as illustrated by the pie charts (Figure 5) indicate a sharp decline in the frequency of impairments related to exceedences of chemical water quality criteria as do the scatter plots (Figure 3 and Figure 4). Remarkably, patterns of disagreement between the use of chemical and biocriteria-based indicators of aquatic life use attainment have remained similar across these time periods. Situations where the biocriteria-based measures find impairment and the water chemistry indicators do not (35.0-36.9%) are much more common than where the water chemistry indicators identify impairment and the biocriteria-based method indicates attainment (7.5-9.2% of stations). Areas of agreement include: 1) where both methods indicate attainment, which changed from 23.1% to 46.5% between the earliest and latest time period and 2) where both methods indicated impairment which changed from 34.4% to 10.5% of stations.

The smallest sections of the pie charts are the sections where water chemistry measures indicate impairment and the biological data indicates attainment. This section of the pie represents one of the most controversial issues related to integrating biological and water chemistry results. The conservative approach to interpreting these results has been called “independent application” and has been generally supported by U.S. EPA in the application of results to the 305(b) and 304(l) sections of the CWA. Absent obvious errors in data, the policy of independent application argues that the scientific basis for both types of monitoring is sound and if collected correctly, any data indicating impairment should be used to list or call a reach impaired even if other data (e.g., biological data) is in disagreement.

[1] Although the is little recent positive trend in habitat condition, attainment of higher tier aquatic life uses are dependent on already existing high quality habitats.

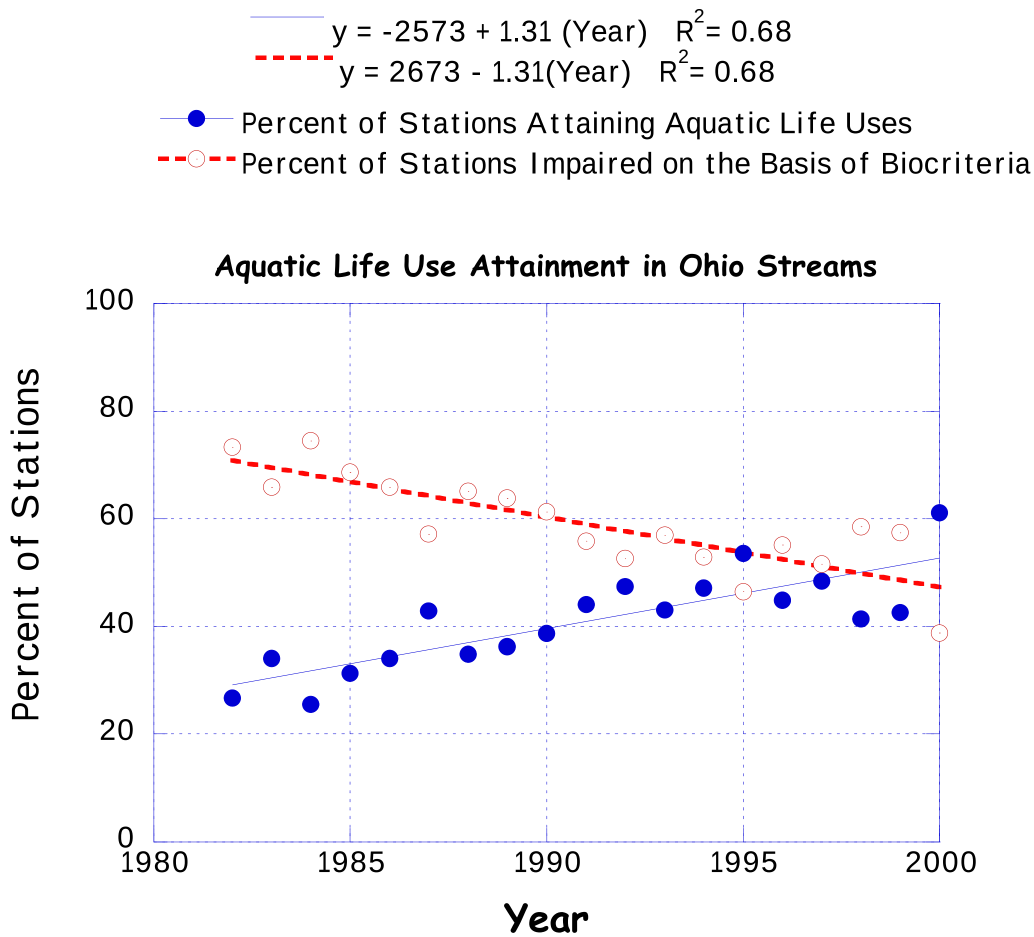


Figure 2. Percent of stations attaining (blue dots) and with impaired (red circles) aquatic life uses in Ohio streams and rivers on the basis of direct assessment with biocriteria from 1982 to 2000 in Ohio streams. Excludes data from the Ohio River and Lake Erie estuary area; data collected by Ohio EPA only. Attainment decisions based on average index values collected from Jun 15-Oct 15 for waters currently designated as EWH, WWH, MWH, LRW, or CWH (see text).

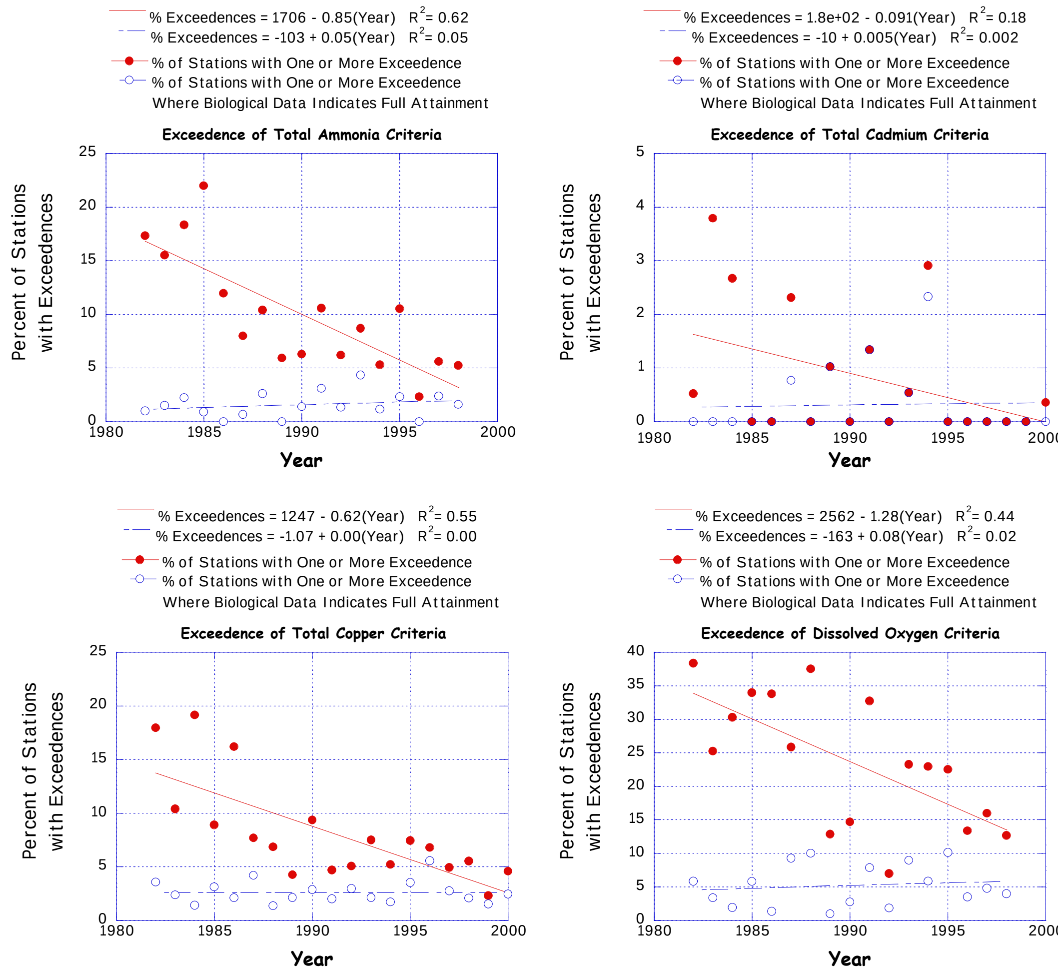
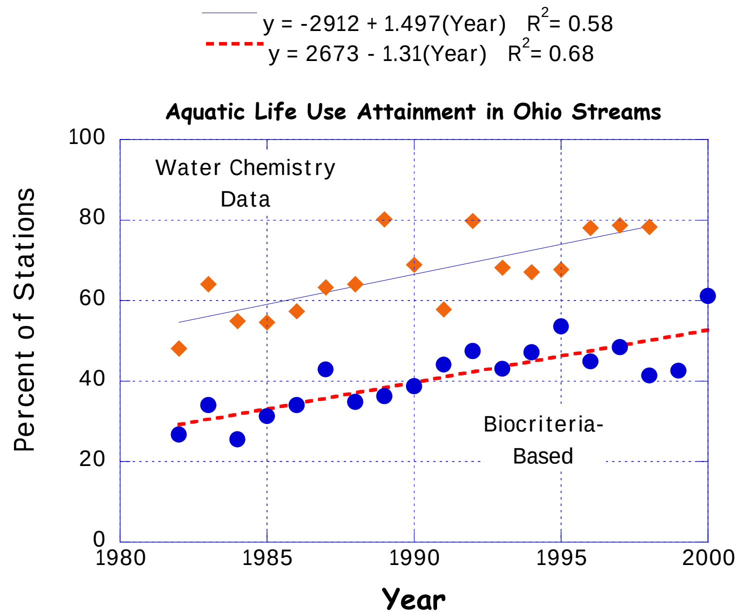


Figure 3. Percent of stations with water quality criteria exceedences (red dots) for total ammonia (top left), total cadium (top right), total copper (bottom left), and dissolved oxygen (bottom right) by year from 1982 to 2000 in Ohio streams. Open dots represent percent of station where there was an exceedence for these compounds, but where biological community data indicated attainment of aquatic life use biological criteria. Only stations with both chemical and biological data collected within the same summer period (June 15-Oct 15) are included.



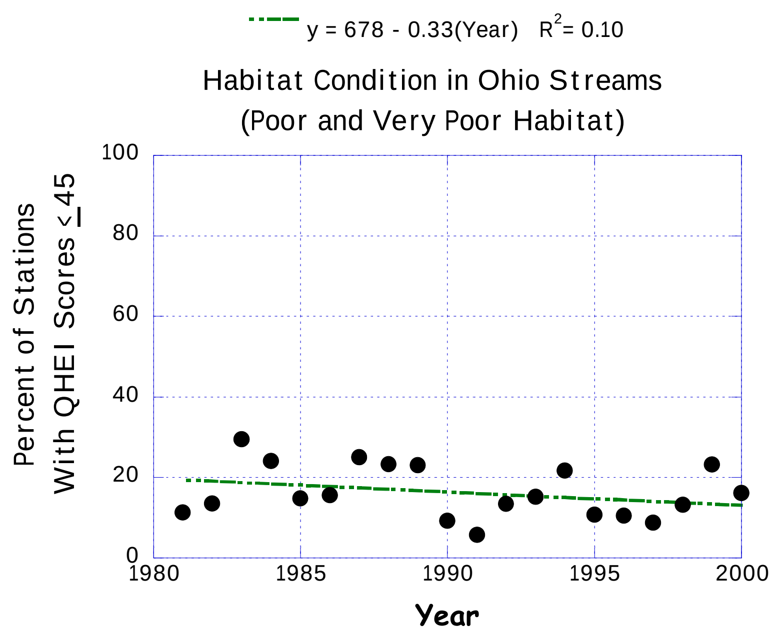


Figure 4. Plots of stations without water chemistry criteria exceedences and percent of stations meeting biocriteria in Ohio streams from 1981-2000 (top panel) and percent sites with poor or very poor habitat quality over time (bottom panel based on the QHEI). Chemical data not complete for 1999 and 2000 and points not represented.

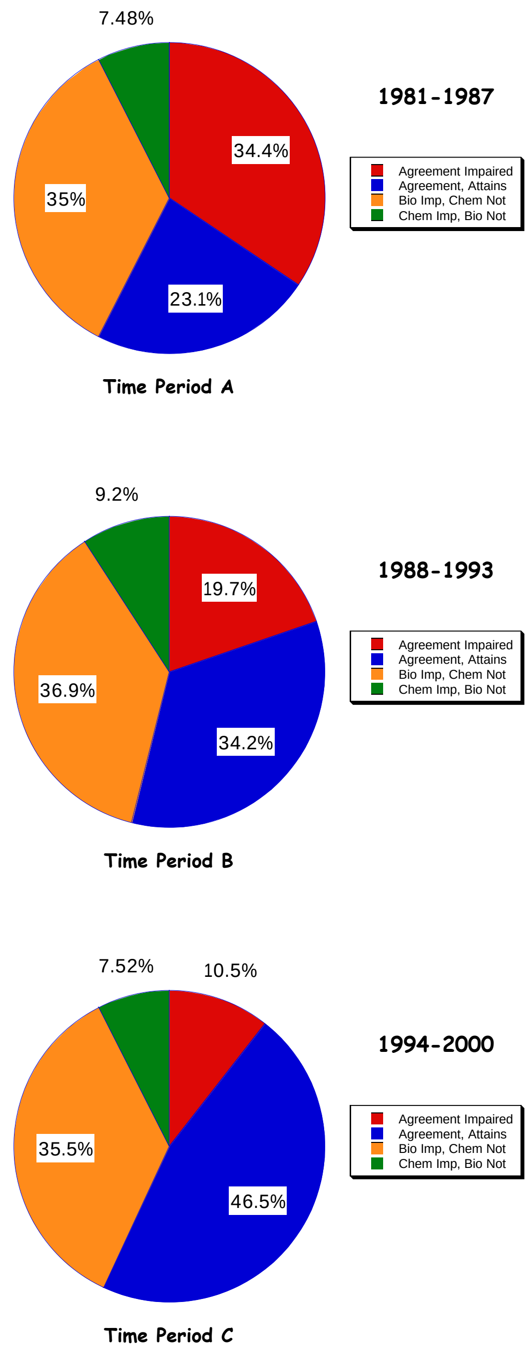


Figure 5. Pie charts summarizing agreement between biocriteria-based vs. water chemistry-based measures of aquatic life use attainment in three different time periods: 1981-1987 (Top), 1988-1993 (Middle), and 1994-2000 (Bottom).

Another approach is called a weight of evidence approach that weighs each piece of evidence to arrive at a decision that is supported by a preponderance of evidence on hand. It does automatically assume that one indicator is correct all of the time, but in reality more weight is often given to biological data where they are appropriate and the most direct measure of the designated use. The “risks” of using one approach or the other has philosophical, regulatory, and scientific or statistical aspects to them. Here we will largely deal with some of the scientific aspects of this issue.

Ohio’s assessment process relies on the biological data as the ultimate arbiters of aquatic life use attainment and to use water column and sediment chemistry data, habitat data, biological signature information and other information (e.g., fish tissue, spills data, biomarker data, discharge records, land use, fish kill reports) where available to identify responsible stressors (causes) and stressor agents (sources). As discussed elsewhere, tiered aquatic life uses and multiple organism groups, standardized sampling methods, and stream size and spatial stratification (e.g., ecoregions) reduce much of the variability in obtaining an accurate assessment of attainment status using biocriteria (Yoder 1995; Yoder and Rankin 1995a,b; Rankin 2003a draft; Rankin 2003b draft). Most sites used in this assessment in Ohio had between 3-6 chemical grab samples per station within a single summer period. For example the mean number of samples/station for total recoverable copper was 4.8. For the independent application part of this discussion then, the “error” rate when comparing methods could decline as more samples provide a more accurate indication of the chemical regime in the stream or actually increase where the decision criteria are stringent (no exceedence over a three year period) and more data is collected.

Several pieces of information from the Ohio database can help provide some insight into the error rate in the scenario where biocriteria indicates attainment and water column chemistry exceedences indicate impairment (IA scenario). The examples of the change in chemical exceedences over time for four parameters illustrated in Figure 3a-d have two lines. The upper lines are the exceedences at sites that also have biocriteria-based impairment. The lower lines represent those sites in the IA scenatio and that make up the green portion of the pie charts in Figure 5; chemical exceedences with biological attainment. All of the curves for the parameters in these graphs show no significant trend in exceedences over time. Given the strong declining trend in exceedences at sites that also show biocriteria-based impairment, I would have expected some similar decline in the sites where biocriteria-based data was unimpaired. Instead the frequency of these exceedences appear more like “statistical noise” with no trend (Figure 3).

Further information related to inconsistencies between the chemical stressors and the biological responses can be derived from examining which parameters were exceeded and to examine the causes identified in the Ohio 305(b) and intensive survey process that brings in data external to the simple water column parameters considered here. Figures 5a-d represent histograms of causes of impairment based on exceedences of chemical water quality criteria or targets (brown bars) or frequency of causes, based on 305(b) assessments (blue bars), associated with impairments in stream reaches that encompass the sites where chemical exceedences were recorded.

The 305(b) assessments utilize the result of intensive survey results and include much more chemical, toxicity, and source data than was included in the simple chemical exceedence exercise that forms the basis of many of the analyses in this paper. Much of this in not easily obtainable in electronic form.

The 305(b) results, because they are summarized by reach (mean length of assessed segments 9.4 miles) suffer from a “spatial indeterminancy” problem -- the impairment described could be actually upstream or downstream of the sampled site. Even so, this data can be useful in exploring which stressors were at least in the proximity of the individual stations and exploring how similar the 305(b) identified stressors were to the parameters that exceeded water quality criteria in our study.

IA Scenario

When we examined all aquatic life uses combined (Figure 6c), exceedences of ecoregion nutrient targets, total metals water quality criteria, and dissolved oxygen criteria accounted for the largest proportion of water chemistry variables exceeded where biocriteria was attained. The nutrient numbers used here are ecoregional and stream-sized based targets derived from observed relationships between biocriteria and observed low-flow nutrient data. We know there can be a substantial amount of variation in the effect a nutrient on a biological endpoint depending on shading, flow, habitat, etc and that while useful as BMP targets, a elevated nutrient number may not always be associated with impairment. The 305(b) summaries use multiple lines of evidence typically based on more detailed data than contained in the water chemistry exceedence analyses we performed here. In figure 6c reach level impairments were less frequently attributed to the effects of elevated nutrient levels than a simple target indicator from a single station.

Dissolved oxygen problems are still a major cause of impairment in Ohio streams and were the third most common major cause behind habitat degradation and siltation in the 2000 Ohio 305(b) report (Ohio EPA 2000). Dissolved oxygen impairment was the most prevalent cause of impairment as recently as 1994, but point source abatement has greatly reduced point sources of this stressor. While dissolved oxygen is an important stressor and related to biological condition, occasional deviations from the criteria may not always be strongly correlated with impairment if excursions are short-term or where small sample-size overestimates the effect of the dissolved oxygen regime. Ohio has taken the approach, especially related to NPS pollution that it is often more accurate and cost effective to trade-off samples of water chemistry data, in favor of biological community samples. The prevalence of non-chemical impacts (e.g., habitat, siltation) and the usefulness of biological signatures to help to confirm water chemistry stressor effects supports this approach. A future project will examine such data at watershed scale to explore how results or conclusions would vary at differing spatial densities and sample sizes of data.

The third most common chemical type exceeded where biocriteria attain aquatic life uses are metals. In this data base, the form of metal measured is the total recoverable form of a metal. These forms may overestimate the magnitude of effects in certain cases, especially where dissolved materials, common in Ohio waters, complex with metals, thereby reducing toxicity. In Figure 6c metals were less frequently identified in 305(b) assessments compared to exceedence measures. This was likely related to a weight of evidence assessment that concluded that these exceedences were infrequent and not biologically significant.

Examination of the more sensitive EWH aquatic life use (Figure 6a) and the less sensitive MWH aquatic life uses (Figure 6b) at sites under the IA scenario, found that nutrients were more frequently cited as important causes of impairment in EWH compared to MWH streams on the basis of 305(b) assessments even though the relative frequency of TP exceedences were similar. MWH streams because of their simplified habitat, harbor taxa less sensitive to elevated nutrients than EWH streams which are characterized by intolerant and sensitive taxa. Many of the taxa sensitivities are related to habitat features (e.g., riffles) found in reference streams that are associated with low nutrients and complex habitats.

Dissolved oxygen was more frequently identified as a cause in 305(b) assessments than as measured exceedences in MWH streams (Figure 6b). This is likely due to unmeasured dissolved oxygen sags not identified in grab samples, but perhaps measured in continuous sampling or inferred by other nearby impacts. In some watersheds, such as the Wabash River watershed in Ohio, dissolved oxygen impacts are strongly associated with manure application which provides a carbon source as well as high nutrient levels. These are generally distinguished in Ohio's database from sites where nutrient runoff may be elevated, but the carbon and other organic matter associated with manure is less common. In EWH streams exceedences and identification of dissolved oxygen impacts from 305(b) assessments were similar (Figure 6a).

In EWH streams there were no sites that were below a QHEI target of 50 which is a transitional score between MWH and WWH streams; this is not unexpected. However, 305(b) assessments found habitat impacts very important in these EWH stream reaches (Figure 6a). Habitat changes can have clear effects in EWH streams at QHEI scores above 50 or much higher when certain important habitat features such as substrate are degraded (e.g., poor substrates in streams with other habitat features remaining intact). In contrast, in MWH streams the proportion of sites with QHEI scores less than 50 were similar to the 305(b) estimates of habitat loss (Figure 6b) suggesting that habitat impacts were more severe and less subtle. In addition, in EWH and MWH streams (Figure 6a,b) flow alteration is a small, but persistent factor in impairment of these waters. Flow alteration is typical associated with drainage and flood control which can result in more frequent drought-like conditions as well more frequent and severe storm flows.

Causes of Impairment Under Other Scenarios

The previous discussion focused on the least common, but most controversial scenario– where water chemistry exceedences occur, but the biocriteria indicate aquatic life use attainment. Much of this controvery is directly related to regulatory and legal issues under the CWA and NPDES program. Other scenarios, however, are of more importance to progress in restoring degraded waters and maintaining conditions in high quality waters largely because 1) they are more prevalent, and 2) they have application to monitoring designs that capture the wide range of stressors, many non-chemical, in addition to water chemistry exceedences.

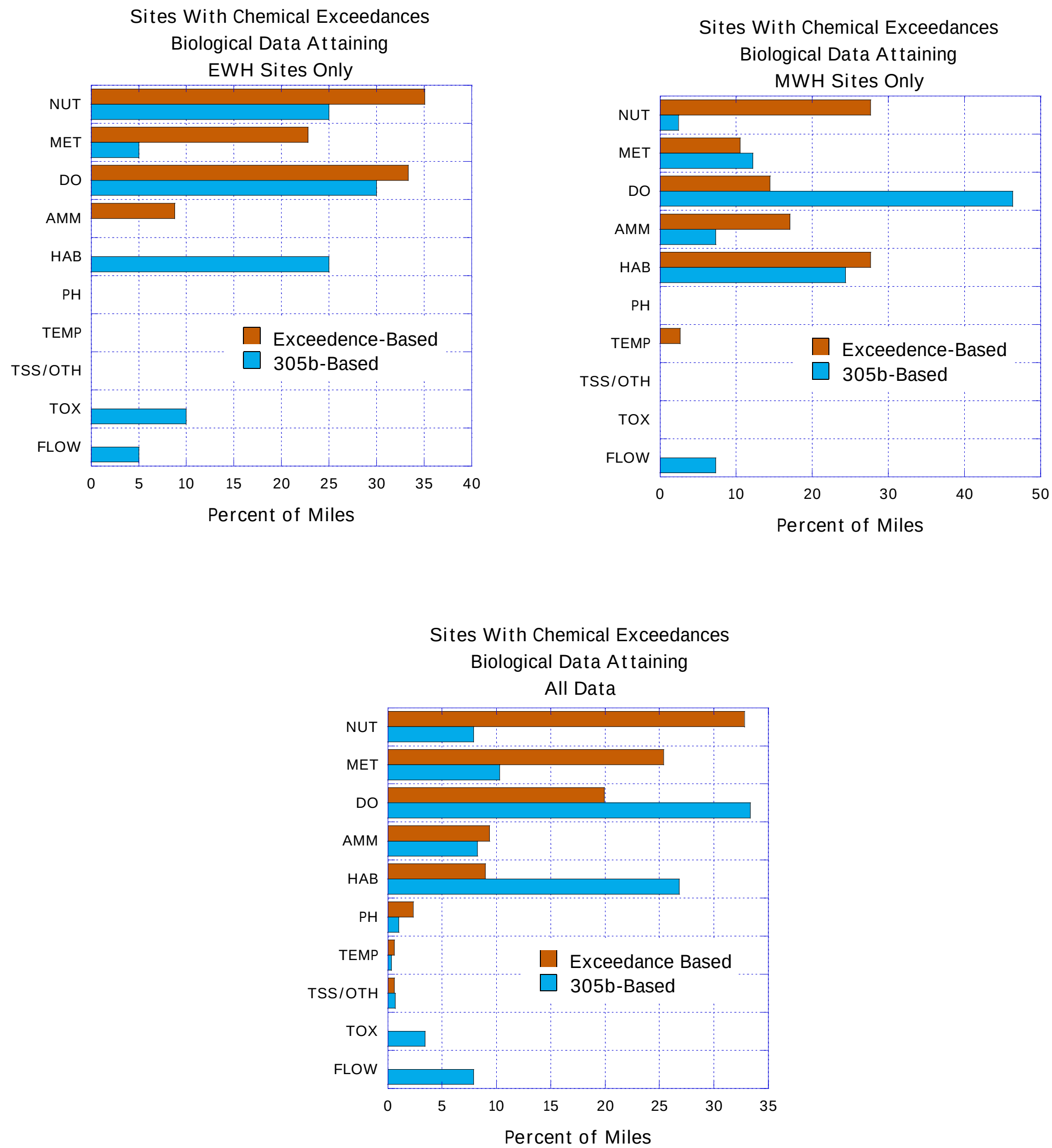


Figure 6a-c. Histograms of causes of aquatic life impairment on the basis of 305(b) assessments by river segment (blue) or by site-specific exceedences of chemical water quality criteria or targets (nutrients) at sites where biocriteria indicated attainment of aquatic use. Numbers reflect relative percent of miles for each cause. (A) Top left (EWH aquatic life use only, (B) Top Right (MWH aquatic life uses only), (C) Bottom (All aquatic life uses).

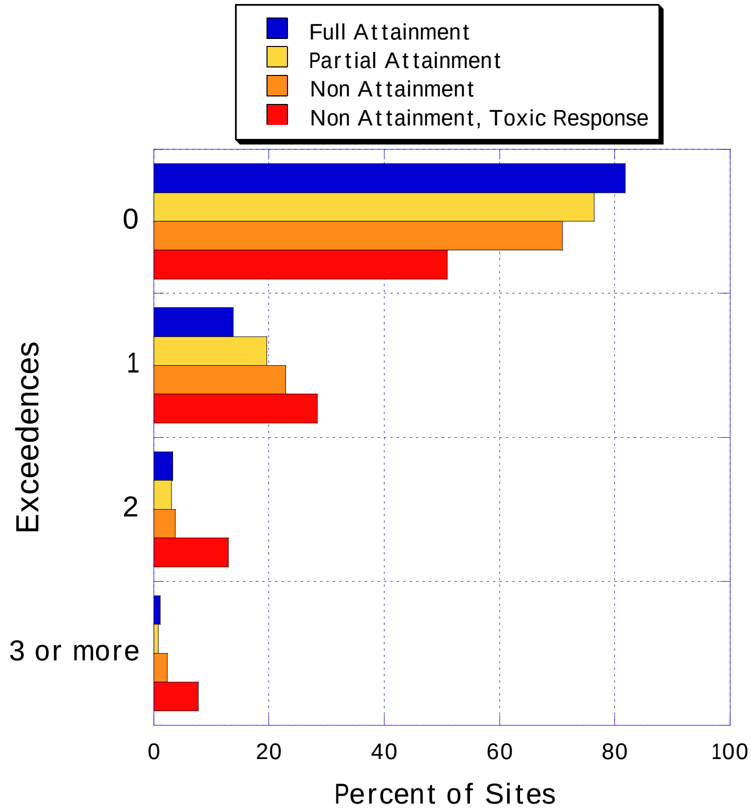


Figure 7. Histogram of the percent sites and the number of parameters that exceeded chemical water quality criteria at each site by biocriteria impairment categories at these sites. Blue indicates full attainment, yellow indicates partial attainment, orange indicates non-attainment and red indicates non-attainment at levels indicating toxic effects.

Other Scenarios
Over all waters and years, the *relative* causes of impairment among the remaining scenarios (biological meeting-chemistry meeting; biology impaired-chemistry impaired; and biology impaired-chemistry meeting), based on 305(b) reach assessments were actually quite similar among all the scenarios except for perhaps the scenario with biological and chemistry being impaired having a greater relative frequency of water chemistry related causes (red bars on Figure 8).

Within each scenario, there were some differences in relative contributions of causes between EWH and MWH uses. For example, across all the scenarios, exceedences of nutrient targets were observed at both EWH and MWH sites, however, reach level, detailed assessments summarized in 305(b) information attributed nutrient impacts to EWH waters only (Figures 9-11). In relation to habitat targets for streams, most indicate degraded habitats for MWH streams, which is not surprising since the use implies limited habitat conditions, but also were reported as important limitations within reaches of these modified streams through 305(b) assessments. This indicates that habitat was so poor as to limit MWH uses or that other uses (WWH, EWH) were part of these segments as well (e.g., upstream or downstream reaches). EWH waters in contrast, did not trigger the QHEI target which is also not surprising since this threshold is designed to detect more severe habitat degradation. The EWH waters however, were identified by 305(b) assessments as being habitat limited as was described for the first scenario above.

Another pattern common across these scenarios is a greater relative contribution of dissolved oxygen impacts based on 305(b) assessments, than from exceedence estimates (Figures 9-11). This can result from more detailed D.O. data available from intensive surveys (e.g., datasondes), other D.O. data not easily available (discharger reported data), or the use of biological response indicators or visual clues to low dissolved oxygen (e.g., decomposing algae mats, obviously anoxic conditions). This implies that areas of agreement between assessment indicators (biology and chemistry both indicating impairment) may well be coincidental in a substantial number of instances. A reliance on a simple, exceedence-based exercise may fortuitously identify an impaired water, but incompletely characterize the nature of the impairment.

Comparing exceedence scenarios side-by-side where biology is attaining vs. where biology is impaired (Figure 12) shows that metal exceedences are relatively more frequent at sites with attaining biocriteria and that exceedences of D.O. and ammonia are relatively more common at sites with impaired biology. Remember that these are relative numbers and that in absolute terms these exceedences are more frequent when both indicator groups are impaired. This greater relative contribution of metals exceedences in unimpaired waters may imply that those exceedences may be less toxic than predicted, plausibly due to complexing with organic matter or small samples sizes inadequately characterizing the chemical regime.

The relationships discussed here can be very complex and we have just provided some plausible explanations about the patterns we have observed. It is clear that no single parameter or group of parameters (e.g., metals) is responsible for all of the differences among indicator results. In fact, the causes identified among all of the various scenarios on the basis of the 305(b) assessments, aside from some EWH vs.s MWH patterns, are fairly similar. This suggests to us that the deviations between chemical and biological indicator arise from strengths and weakness related to their roles as response (biological data) or stressor (chemical data) indicators. There does not appear to be a single “smoking gun” parameter that explains differences between theses indicator groups. We think this data supports an integrated monitoring approach where each indicator tools is used in its proper role. Inappropriately using water chemistry data as a response indicator may result in large errors of omission by 1) failing to detect impacts from non-chemical (e.g., habitat) or 2) misclassifying the nature of the impact (coincidental impairments). Similarly, biological data by itself is insufficient to characterize specific stressors needed to decide on abatement strategies, even though categories of stressor may be identified. Finally biological data is well suited to estimate severity of impairments that would be difficult or costly to do based on chemical stressors alone.

Independent Application Issues

These results do effect consideration of independent application, especially for ambient assessments of condition. We see this as different from use in specific permit of enforcement issues where there application may be used independently to ensure a large safety factor. For ambient assessment purposes, however, use of independent application may institutionalize more simplified approaches to examining aquatic life use attainment/impairment issues. If a more complete data set does not allow more measured, risk-based approaches to detecting impairment, collection of “additional” data is seen as a burden instead of a tool to arrive at the most accurate assessment of condition. Disincentives to collect and broadly integrate biocriteria into state programs are important because the risks of not “listing” chemically impaired sites is dwarfed by the 1) failure to identify waters as impaired because bioassessment data is absent, and 2) identifying a water as impaired correctly, but miscategorizing the cause of the impairment.

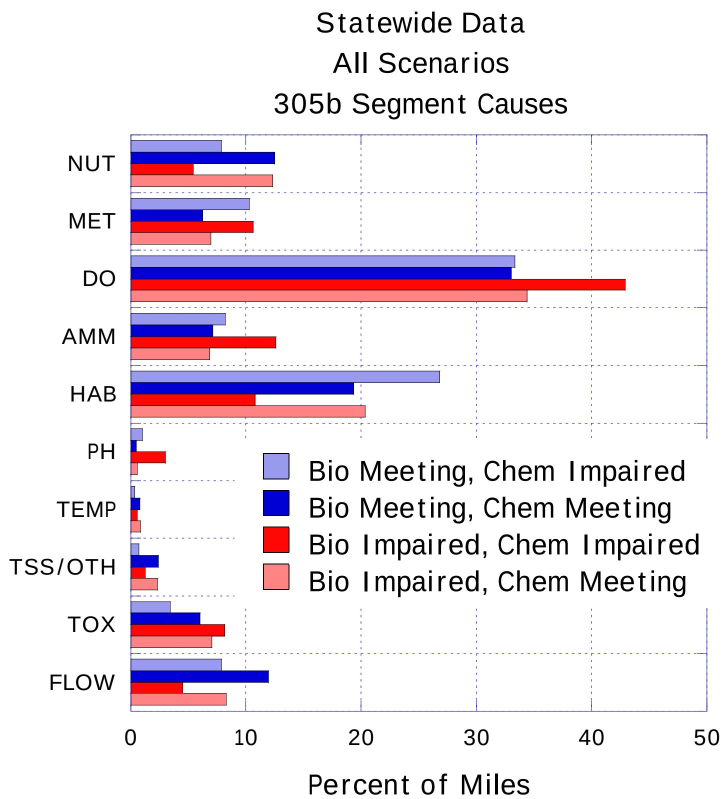
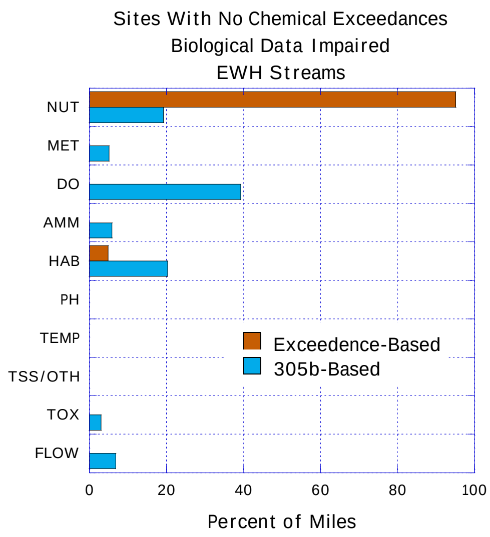
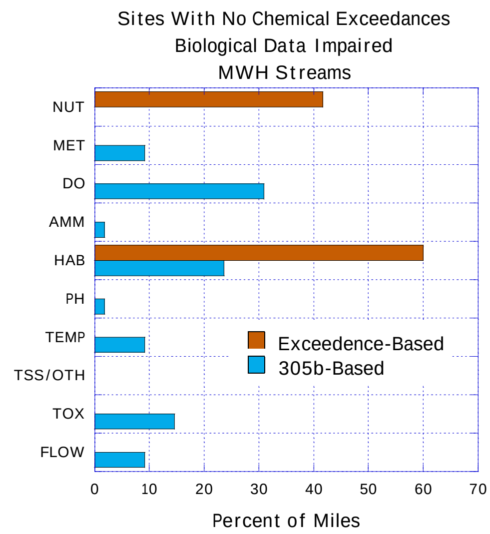


Figure 8. Histogram of causes of aquatic life impairment on the basis of 305(b) reach-level assessments of stressors from intensive survey data where: 1) sites are meeting biocriteria, but water chemistry indicates impairment (blue hatched); 2) sites are meeting biocriteria and water chemistry criteria (solid blue); 3) sites where biocriteria is impaired and water chemistry indicates impairment(solid red); and 4) sites where biocriteria is impaired and there are no exceedences of water chemistry criteria (red hatched). Numbers reflect relative percent of miles for each cause. All aquatic life use combined.





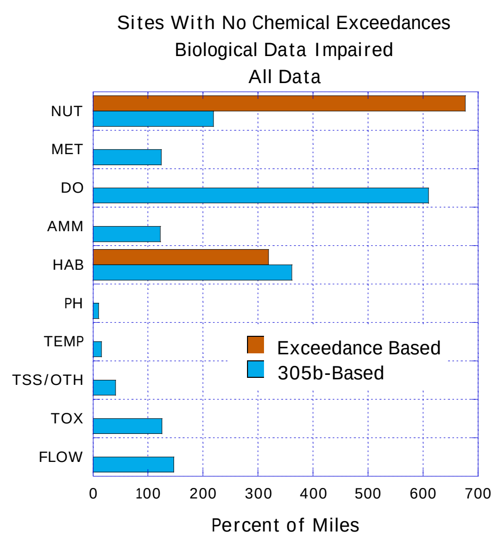


Figure 9a-c. Histograms of causes of aquatic life impairment on the basis of 305(b) assessments by river segment (blue) or by site-specific exceedences of chemical water quality criteria or targets (nutrients) at sites where biocriteria indicated impairment of aquatic use. Numbers reflect relative percent of miles for each cause. (A) Top left (EWH aquatic life use only, (B) Top Right (MWH aquatic life uses only), (C) Bottom (All aquatic life uses).

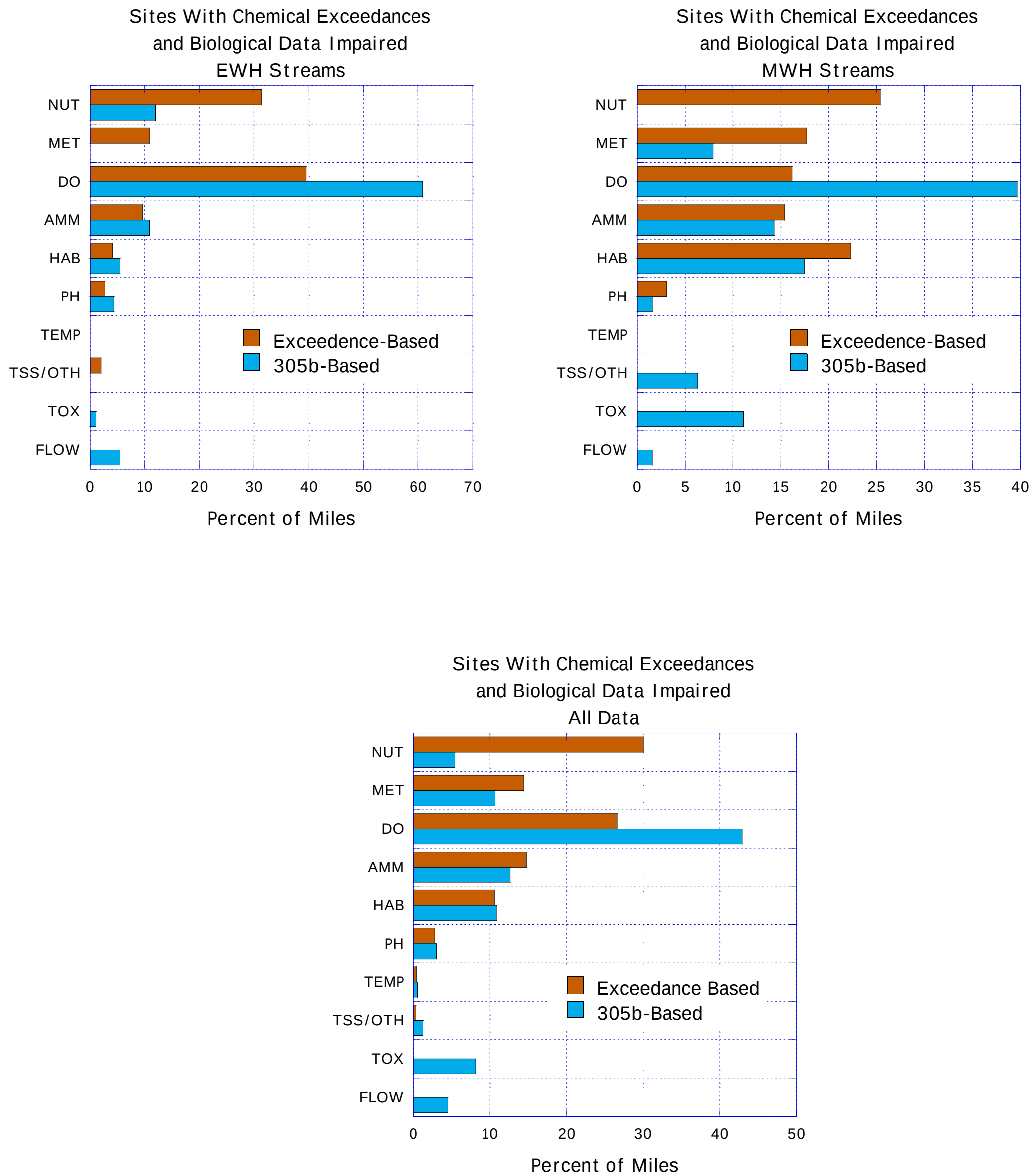


Figure 10a-c. Histograms of causes of aquatic life impairment on the basis of 305(b) assessments by river segment (blue) or by site-specific exceedances of chemical water quality criteria or targets (nutrients) at sites where biocriteria and water quality criteria indicated impairment of aquatic use. Numbers reflect relative percent of miles for each cause. (A) Top left (EWH aquatic life use only, (B) Top Right (MWH aquatic life uses only), (C) Bottom (All aquatic life uses).

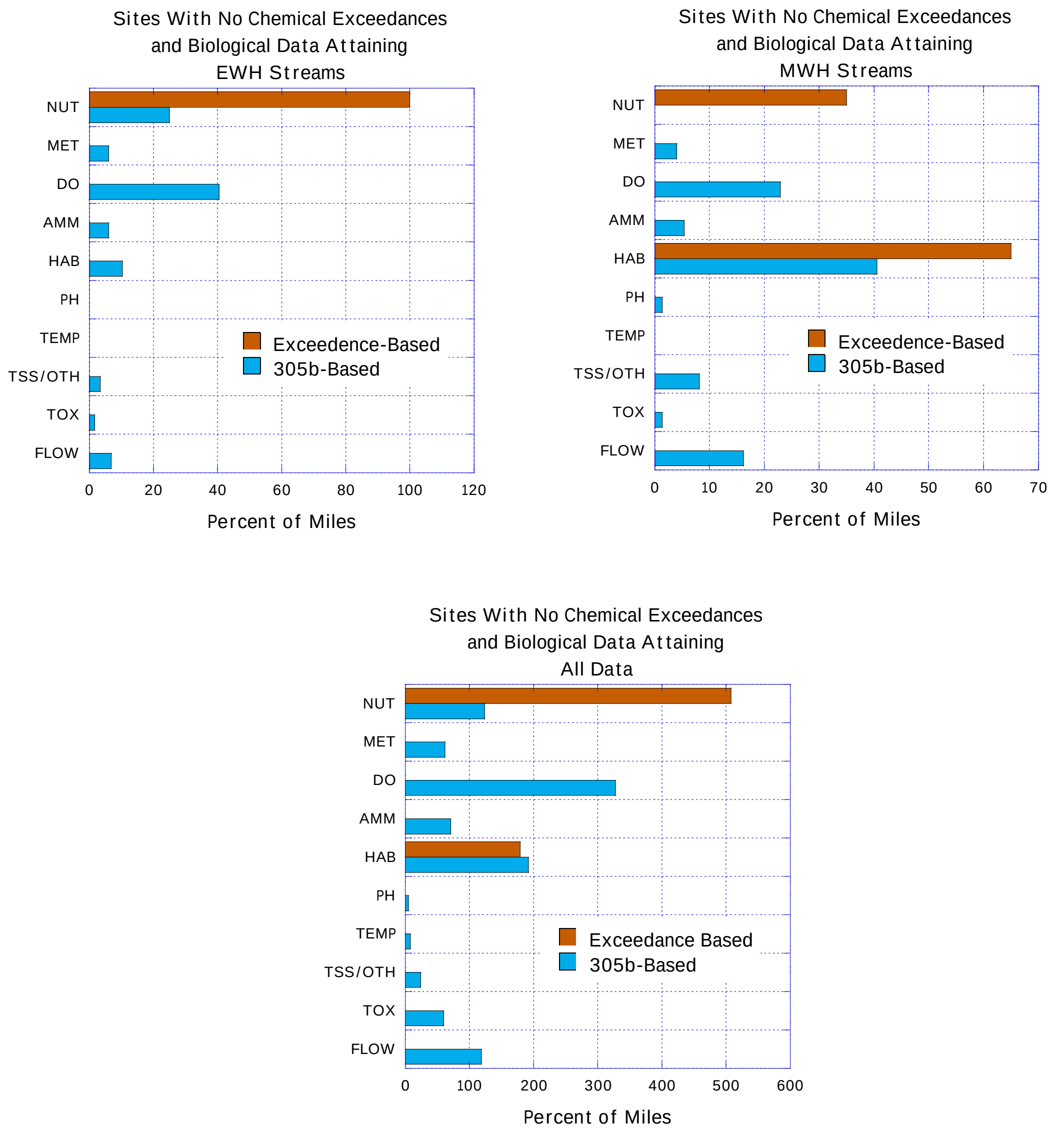


Figure 11a-c. Histograms of causes of aquatic life impairment on the basis of 305(b) assessments by river segment (blue) or by site-specific exceedences of chemical water quality criteria or targets (nutrients) at sites where biocriteria and water quality criteria indicated attainment of aquatic life uses. Numbers reflect relative percent of miles for each cause. (A) Top left (EWH aquatic life use only, (B) Top Right (MWH aquatic life uses only), (C) Bottom (All aquatic life uses).

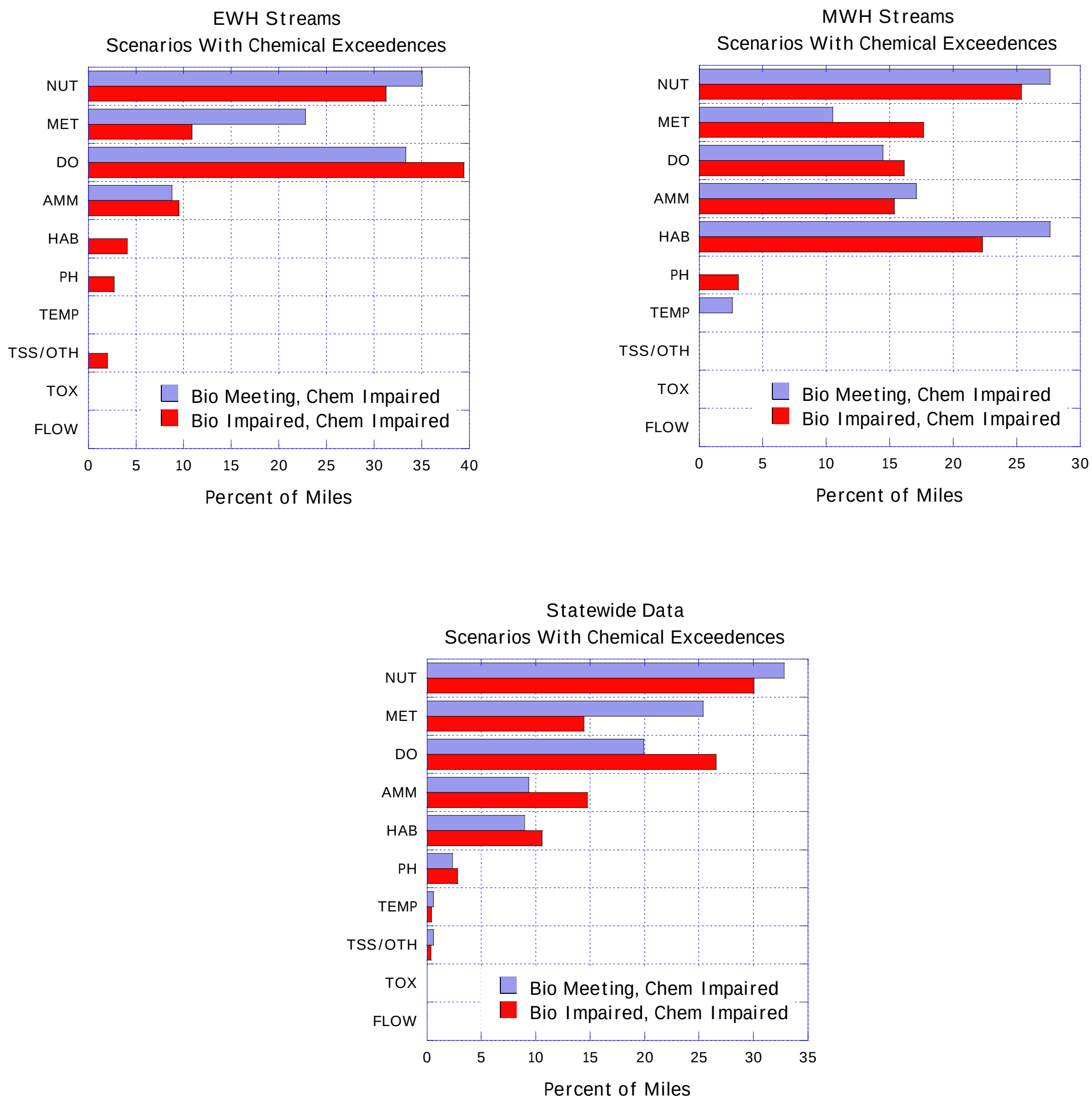


Figure 12a-c. Histograms of causes of aquatic life impairment on the basis of site-specific exceedences of chemical water quality criteria or targets (nutrients) at sites where biology is meeting biocriteria (blue hatched) or where biocriteria is impaired (solid red). Numbers reflect relative percent of miles for each cause. (A) Top left (EWH aquatic life use only, (B) Top Right (MWH aquatic life uses only), (C) Bottom (All aquatic life uses).

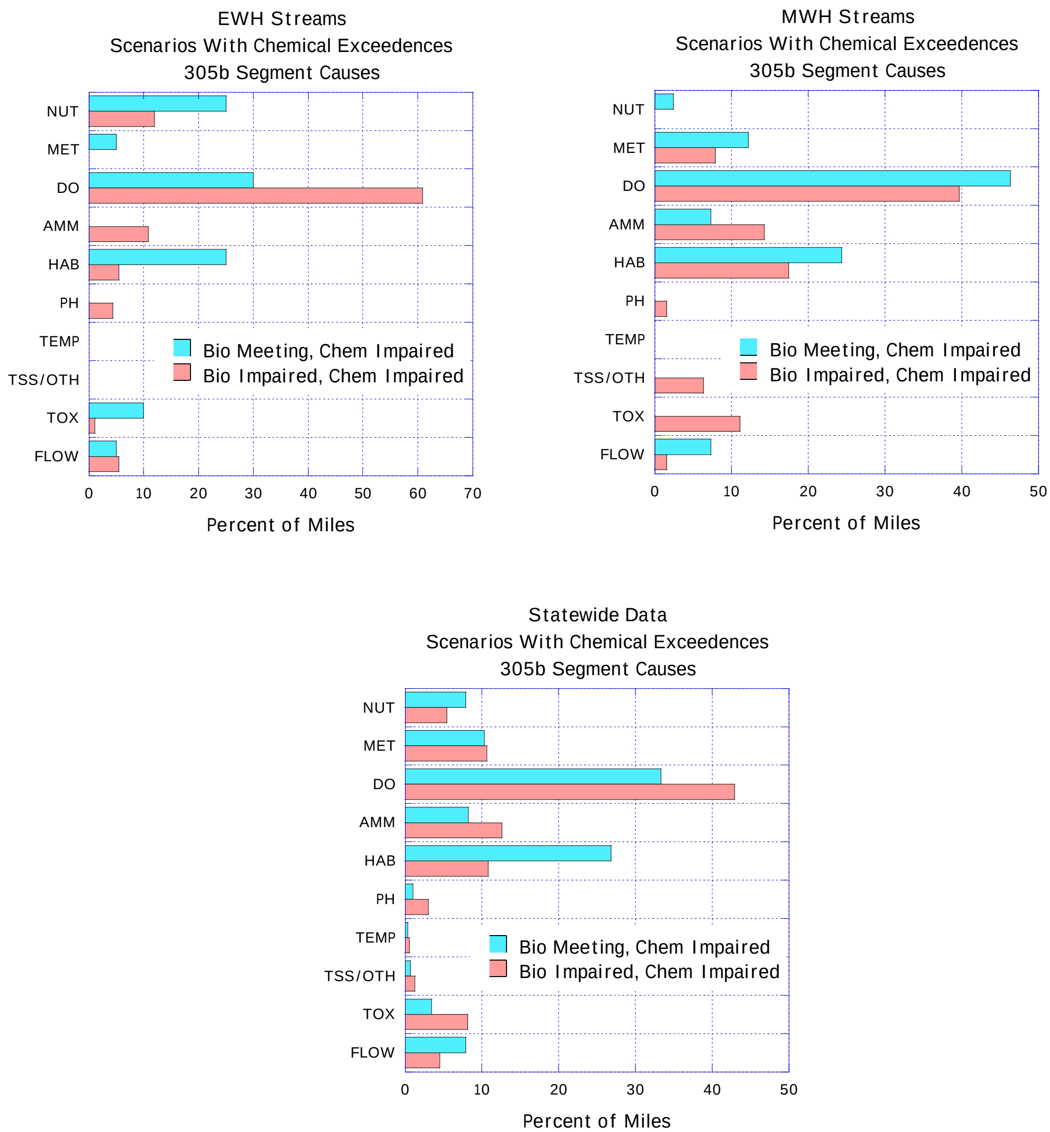


Figure 13a-c. Histograms of causes of aquatic life impairment on the basis of 305(b) reach-level assessments of stressors from intensive survey data where biology is meeting biocriteria (blue solid) or where biocriteria is impaired (hatched red). Numbers reflect relative percent of miles for each cause. (A) Top left (EWH aquatic life use only, (B) Top Right (MWH aquatic life uses only), (C) Bottom (All aquatic life uses).

Conclusion
This study confirmed that for Ohio and likely other Midwest states, biocriteria in an integrated intensive survey framework is the most powerful way to identify attainment and impairment of aquatic life uses. This follows the arguments by the National Academy of Science TMDL committee (National Research Council 2001) that indicators that most directly measure a designated use will be the most accurate indicators. As was shown in a segment analyses of 305(b) data in Ohio from 1996, the proportion of situations where water chemistry data indicate impairment, but biocriteria indicate attainment are infrequent and have not changed little over time, even as water quality has improved. In contrast a simple, water quality exceedence approach to detecting impairment failed to detect impairment in more than 1/3 of the stations. Analysis of causes of impairment from intensive survey results suggest that some agreement about impairment between biocriteria and water chemical approaches is coincidental. General consistency of causes among the scenarios we examined suggests that no single parameter was responsible for conflicting results and that this implies that for ambient monitoring it is important to use indicators in their most effective role, whether that is as a response indicator or as a stressor indicator.

This study, paired with companion analyses examining strengths of 1) using muiltiple organism groups (Rankin 2003a Draft) and 2) using tiered aquatic life uses (Rankin 2003b Draft) suggests that extensive variation in measuring impairment can be greatly reduced using an adequate monitoring approach focused on biological endpoints with the ability to use multiple organism groups and to have tiered expections for waterbodies. States often cut monitoring programs first during budget crises, however, the effect on the public and the regulated community could potentially be large. U. S. EPA draft cost estimates of implementing TMDLs ranged up to 4.3 billion dollars annually (U. S. EPA 2001). State monitoring programs that are incomplete and poorly based could have error rates of 30% or more based on this study and may be prone to identifying an incorrect or incomplete characterization of responsible stressors. A more complete analysis of the costs related to improperly classifying the extent and nature of impairments would be a useful tool for those prioritizing water quality management efforts.

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