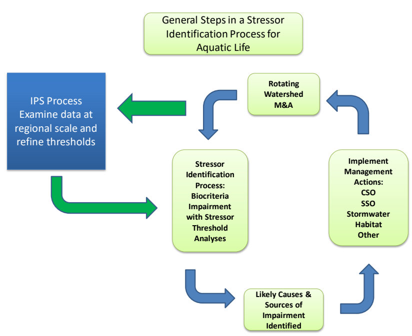
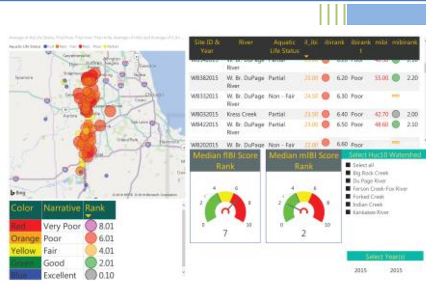
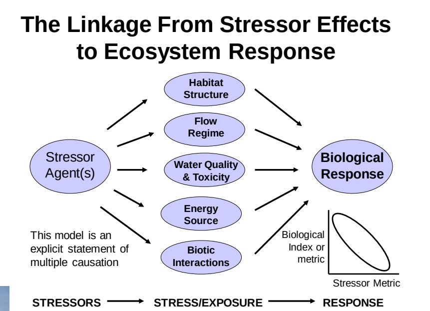
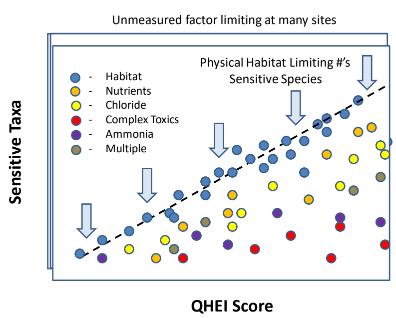
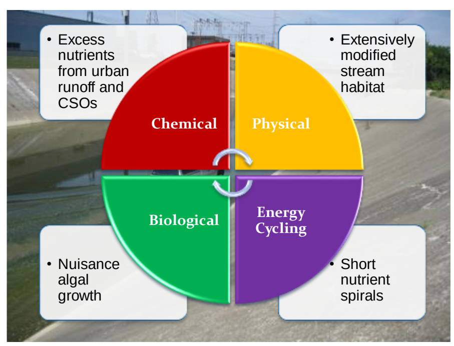
# Integrated Prioritization System (IPS) for Northeastern Illinois: Technical Documentation and Atlas of Stressor Relationships

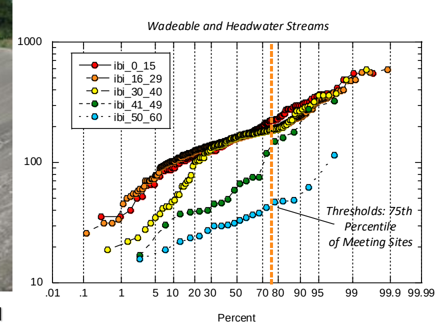


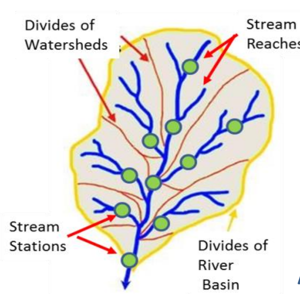



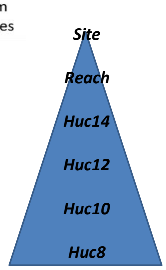


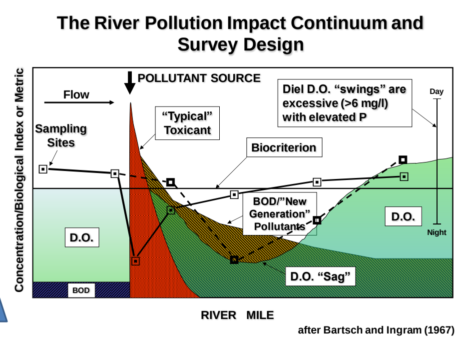



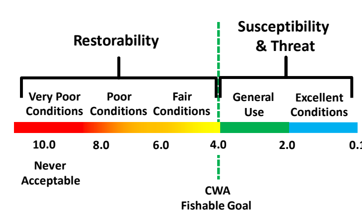

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# Integrated Prioritization System (IPS) for Northeastern Illinois: Technical Documentation and Atlas of Stressor Relationships

MBI Technical Report MBI/2020-5-10

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## Northeastern Illinois Integrated Prioritization System (IPS) Disclaimer

The Northeastern Illinois Integrated Prioritization System (IPS) provides a framework within which high resolution monitoring data and assessment results are organized, analyzed, and merged to better support Clean Water Act (CWA)[1] management programs in meeting their goals and objectives and guiding a wide array of water quality related decision-making. It was developed to better inform the actions of watershed groups in Northeastern Illinois by identifying patterns and thresholds of stressors that affect aquatic life condition based on comprehensive analyses of a regional database comprised of high resolution biological, habitat, chemical/physical, and land use parameters and indicators. The data was generated by the systematic watershed monitoring conducted by the DuPage River Salt Creek Workgroup (DRSCW) since 2006, the Lower DuPage River Watershed Coalition (LDRWC) since 2012, the Des Plaines River Watershed Workgroup (DRWW) since 2016, and the North Branch Chicago River Watershed Workgroup (NBWW) since 2018 that has been focused on determining the status of Illinois aquatic life designated uses and determining the causes (agents) and sources (origins) of impairments and threats. Suitable data from Illinois EPA and Illinois DNR was also used to supplement these more spatially intensive datasets. The limits of application include small headwater streams, wadeable streams, and small rivers up to 350 mi.$^2$ in drainage area. The thresholds and analyses are not suitable for application outside of these stream and river sizes.

The IPS houses analyses of complex environmental data about biological indicators and the effect that chemical, physical, and land use variables have on the measured and potential condition of the biota and water quality at the site, stream or river reach, and watershed (HUC12) scales. Regionally scoped data of sufficient resolution is required to determine the extent and severity of stream and river impairments and for developing stressor thresholds for supporting local scale restoration and protection. The IPS is intended to be used where systematic monitoring and assessment has been conducted at the local watershed scale of resolution (e.g., HUC12) such that paired biological, chemical, physical, and land use data is generated to evaluate current conditions. In places where existing data is spatially insufficient, a systematic collection of data to fill spatial gaps will need to be undertaken as a first step towards IPS usage.

The stressor thresholds generated by the IPS do not constitute Water Quality Standards (WQS) hence the members of the funding organizations have no statutory obligation to use them for specific regulatory or non-regulatory actions. However, making quantitative indicators and tools available to guide and support restoration and protection efforts undertaken by local watershed groups and their respective stakeholders is a primary purpose of the IPS framework. It is purposed to provide supporting data and information to restore impaired streams and rivers and also to protect high quality sites, reaches, and watersheds from further degradation. Users should select and interpret the content and information provided by the IPS within the context of how they should be applied to individual watersheds for supporting existing and future programs and obligations and the potential to collaborate with their watershed partners.

[1] Federal Water Pollution Control Act, Public Law 92-500 (33 U.S.C. 1251 et seq.) as amended via PL 107-303, Nov. 2002.

## ACKNOWLEDGEMENTS

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## Glossary of Terms

**Ambient Monitoring** — Sampling and evaluation of receiving waters not necessarily associated with episodic perturbations.

**Aquatic Assemblage** — An association of interacting populations of organisms in a given waterbody, for example, the fish assemblage or the benthic macroinvertebrate assemblage.

**Aquatic Community** — An association of interacting assemblages in a given waterbody, the biotic component of an ecosystem.

**Aquatic Life Use (ALU)** — 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 custodial State agency.

**Assemblage** — Refers to all of the various species of a particular taxonomic grouping (e.g., fish, macroinvertebrates, algae, submergent aquatic plants, etc.) that exist in a particular habitat. Operationally this term is useful for defining biological assessment methods and their attendant assessment mechanisms, i.e., indices of biotic integrity (IBI), O/E models, or fuzzy set models.

**Attainment Status** — The state of condition of a waterbody as measured by chemical, physical, and biological indicators. Full attainment is the point at which measured indicators signify that a water quality standard has been met and it signifies that the designated use is both attained and protected. Non-attainment is when the designated use is not attained based on one or more of these indicators being below the required condition or state for that measure or parameter.

**Attribute** — A measurable part or process of a biological system.

**Benchmark** — A standard or point of reference by which a receiving waterbody chemical, physical, or biological result is measured and judged.

**Beneficial Uses** — Desirable uses that acceptable water quality should support. Examples are drinking water supply, primary contact recreation (such as swimming), and aquatic life support.

**Benthic Macroinvertebrates** — Animals without backbones, living in or on the substrates, 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 (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, pollution, or stressor effect.

**Biological Assessment** — An evaluation of the biological condition of a waterbody using surveys of the structure and function of a community of resident biota; also known as bioassessment. It also includes the interdisciplinary process of determining condition and relating that condition to chemical, physical, and biological factors that are measured along with the biological sampling.

**Biological Criteria (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; also known as biocriteria.

Regulatory meaning: narrative descriptions or numerical values of the structure and function of aquatic communities in a waterbody necessary to protect a designated aquatic life use, implemented in, or through state water quality standards.

**Bioassessment Based Approach** — This approach includes refined aquatic life uses (ALUs) based on numeric biological criteria and implementation via an adequate monitoring and assessment program that includes biological, chemical, and physical measures, parameters, indicators and a process for stressor identification.

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| **Biological Condition Gradient** | A scientific model that describes the biological responses within an aquatic ecosystem to the increasing effects of stressors. |
| **Biological Diversity** | 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 taxa and their relative frequencies. For biological diversity, these taxa 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; also known as biodiversity. |
| **Biological Indicator** | An organism, species, assemblage, or community characteristic of a particular habitat, or indicative of a particular set of environmental conditions; also known as a bioindicator. |
| **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 (after Karr and Dudley 1981). |
| **Biological Monitoring** | The use of a biological entity (taxon, species, assemblage) as a detector and its response as a measure of response to determine environmental conditions. Ambient biological surveys and toxicity tests are common biological monitoring methods; also known as biomonitoring. |
| **Biological Survey** | The collection, processing, and analysis of a representative portion of the resident aquatic community to determine its structural and/or functional characteristics and hence its condition using standardized methods. |
| **Clean Water Act (CWA)** | 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.; referred to herein as the CWA. |

**CWA Section 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.

**CWA Section 305(b)**
Biennial reporting required by the Act to describe the quality of the Nation's surface waters, to serve as an evaluation of progress made in maintaining and restoring water quality, and describe the extent of remaining problems.

**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 and are commonly expressed as a chemical concentration, a physical parameter, or a biological assemblage endpoint.

**DELT Anomalies**
The percentage of Deformities, Erosions (e.g., fins, barbels), Lesions and Tumors on fish assemblages (DELT). An important fish assemblage attribute that is a commonly employed metric in fish IBIs.

**Designated Uses**
Those uses specified in state water quality standards for each waterbody or segment whether or not they are being attained.

**Disturbance**
Any activity of natural or human causes that alters the natural state of the environment and its attributes and which can occur at or across many spatial and temporal scales.

**Ecological integrity**
The summation of chemical, physical, and biological integrity capable of supporting and maintaining a balanced, integrated adaptive community of organisms having a species composition, diversity, and functional

organization comparable to that of natural habitats in the region.

**Ecoregion** A relatively homogeneous geographical area defined by a similarity of climate, landform, soil, potential natural vegetation, hydrology, or other ecologically relevant variables; ecoregions are portioned at increasing levels of spatial detail from level I to level IV.

**Existing Use** A use that was actually attained in a waterbody on or after November 28, 1975, whether or not they are included in the state water quality standards (November 28, 1975 is the date on which U.S. EPA promulgated its first water quality standards regulation in 40CFR Part 131). Existing uses must be maintained and cannot be removed.

**Index of Biotic Integrity (IBI)** An integrative expression of site condition across multiple metrics comprised of attributes of a biological assemblage. It refers to the index developed by Karr (1981) and explained by Karr et al. (1986). It has been used to express the condition of fish, macroinvertebrate, algal, and terrestrial assemblages throughout the U.S. and in each of five major continents.

**Modified Index of Well-Being (MIwb)** The Modified Index of Well-Being (MIwb) is based on fish assemblage measures including numbers, biomass, and two diversity indices (Shannon Index) based on numbers and biomass. The numbers and biomass metrics exclude highly tolerant species. It reflects the overall productivity and diversity of the fish assemblage and it frequently responds before the IBI to improvements in water quality and habitat.

**Metric** A calculated term or enumeration representing an attribute of a biological assemblage, usually a structural aspect, that changes in a predictable manner with an increased effect of human disturbance.

**Monitoring and Assessment** The entire process of collecting data from the aquatic environment using standardized methods and protocols, managing that data, analyzing that data to make assessments in support of multiple program objectives, and disseminating the assessments to stakeholders and the public.

**Multimetric Index** An index that combines assemblage attributes, 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.

**Narrative Biocriteria** Written statements describing the narrative attributes of the structure and function of aquatic communities in a waterbody necessary to protect a designated aquatic life use.

**Natural Condition** This includes the multiplicity of factors that determine the physical, chemical, or biological conditions that would exist in a waterbody in the absence of measurable impacts from human activity or influence.

**Numeric Biocriteria** Specific quantitative and numeric measures of the structure and function of aquatic communities in a waterbody necessary to protect a designated aquatic life use.

**Qualitative Habitat Evaluation Index** A qualitative habitat evaluation assessment tool that is applied to streams and rivers in Ohio and which is used to identify habitat variables that are important to attainment of the Ohio biological criteria.

**Reference Condition** The condition that approximates natural, unimpacted to best attainable conditions (biological, chemical, physical, etc.) for a waterbody. Reference condition is best determined by collecting measurements at a number of sites in a similar waterbody class or region under minimally or least disturbed conditions (by human activity), if they exist. Since undisturbed or minimally disturbed conditions may be difficult or impossible to find in some states, 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 comprehended. Reference condition

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| | is used as a benchmark to establish numeric biocriteria. |
| **Reference Site** | A site selected to represent an approximation of reference condition and by comparison to other sites being assessed. For the purpose of assessing the ecological condition of other sites, a reference site is a specific locality on a waterbody that is minimally or least disturbed and is representative of the expected ecological condition of other localities on the same waterbody or nearby waterbodies. |
| **Refined Aquatic Life Uses** | As defined: The structure of designated aquatic life uses that incorporates a hierarchy of use subclasses and stratification by natural divisions that pertain to geographical and waterbody class strata. Refined ALUs are based on representative ecological attributes and these should be reflected in the narrative description of each ALU subcategory and be embodied in the measurements that extend to expressions of that narrative through numeric biocriteria and by extension to chemical and physical indictors and criteria.<br><br>As used: Refined ALUs are assigned to water bodies based on the protection and restoration of ecological potential. This means that the assignment of an ALU subcategory to a specific waterbody is done with regard to reasonable restoration or protection expectations and attainability. Hence knowledge of the current condition of a waterbody and an accompanying and adequate assessment of stressors affecting that waterbody are needed to make these assignments. |
| **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, subregion, bioregion, or major drainage unit. |
| **Stressors** | Physical, chemical, and biological factors that can adversely affect aquatic organisms. The effect of stressors is apparent in the biological responses. |

**Use Attainability Analysis (UAA)** A structured scientific assessment of the physical, chemical, biological or economic factors affecting attainment of the uses of waterbodies.

**Use Classes** A broad capture of a designated use for general purposes such as recreation, water supply, and aquatic life.

**Use Subclasses** A subcategorization of use classes into discrete and meaningful descriptions. For aquatic life this would include a hierarchy of warmwater and cold water uses and additional stratification provided by different levels of warmwater uses and further stratification by waterbody types.

**Threshold** The measurable point at which an effect becomes evident in an ambient biological response, i.e., the concentration or otherwise measured level or quantity of a particular chemical, physical, or land use stressor corresponding to a change in a biological assemblage measure.

**Total Maximum Daily Load (TMDL)** The maximum amount of a pollutant that a body of water can receive while still meeting water quality standards. Alternatively, a TMDL is an allocation of a water pollutant deemed acceptable to attain the designated use assigned to the receiving water.

**Water Quality Standards (WQS)** 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 Quality Management** A collection of management programs relevant to a water resource protection that includes problem identification, the need for and placement of best management practices, pollution abatement actions, and measuring the effectiveness of management actions.

## List of Acronyms

| | |
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| **ALU** | Aquatic Life Use |
| **BCG** | Biological Condition Gradient |
| **CMAP** | Chicago Metropolitan Agency for Planning |
| **CWA** | Clean Water Act |
| **DRSCW** | DuPage River Salt Creek Workgroup |
| **DRWW** | Des Plaines River Watershed Workgroup |
| **EPT** | Ephemeroptera, Plecoptera, Trichoptera |
| **FIT** | "Goodness-of-fit" or FIT score |
| **IBI** | Index of Biotic Integrity for fish assemblages |
| **IC** | Impervious Cover |
| **ICI** | Invertebrate Community Index |
| **IEPA** | Illinois Environmental Protection Agency |
| **ITFM** | Intergovernmental Task Force on Monitoring Water Quality |
| **LDWG** | Lower Des Plaines Watershed Group |
| **LDRWC** | Lower DuPage River Watershed Coalition |
| **M&A** | Monitoring and Assessment |
| **MBI** | Midwest Biodiversity Institute |
| **NARP** | Nutrient Assessment Reduction Plan |
| **NBWW** | North Branch Chicago River Watershed Workgroup |
| **NIP** | Nutrient Implementation Plan |
| **NARP** | Nutrient Assessment Reduction Plan |

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| **NLC** | National Land Cover |
| **NLRS** | Nutrient Loss Reduction Strategy |
| **NPDES** | National Pollutant Discharge Elimination System |
| **NRI** | Nutrient Ranking Index |
| **NSAC** | State of Illinois Nutrient Science Advisory Committee |
| **OEPA** | Ohio Environmental Protection Agency |
| **QHEI** | Qualitative Habitat Evaluation Index |
| **SSD** | Sensitive Species Distribution |
| **TMDL** | Total Maximum Daily Load |
| **UAA** | Use Attainability Analysis |
| **WLA** | Waste Load Allocation |
| **WQS** | Water Quality Standards |
| **WSV** | Weighted Stressor Value |
| **WWTP** | Wastewater Treatment Plant |

# Integrated Prioritization System (IPS) for Northeastern Illinois: Technical Documentation and Atlas of Stressor Relationships

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## Chapter 1: Rationale and Background

### INTRODUCTION

The Integrated Prioritization System (IPS) provides a framework within which high resolution monitoring data and assessment results can be organized, analyzed, and merged to better support Clean Water Act (CWA)[2] management programs in meeting their goals, and objectives and guiding a wide array of related water quality decision-making. It is the objective of the Clean Water Act (CWA) to protect and restore the chemical, biological and physical integrity of the Nation's waters (Section 101[a]). To achieve this objective national goals were established by the CWA. Perhaps the most well-known is the goal that *"wherever attainable, an interim goal of water quality which provides for the protection and propagation of fish, shellfish, and wildlife and provides for recreation in and on the water (Section 101[a][2])".* This is commonly referred to as the "fishable/swimmable" goal of the CWA and it provides the legislative foundation for Water Quality Standards (WQS) that are used to measure and manage water quality via monitoring and assessment and water quality-based regulation of sources of pollution. A WQS consists of the designated use and chemical, physical, and biological criteria designed to protect that use. Designated uses broadly include the protection of aquatic life, recreation in and on the water, aesthetics, providing safe water supplies, and consumption uses for protecting humans and wildlife. Both the attainability and attainment of WQS is preferably determined via adequate monitoring and assessment.

The systematic watershed monitoring carried out by the DuPage River Salt Creek Workgroup (DRSCW) since 2006, the Lower DuPage River Watershed Coalition (LDRWC) since 2012, and the Des Plaines River Watershed Workgroup (DRWW) since 2016, and the North Branch Chicago River Watershed Workgroup (NBWW) since 2018 has focused primarily on determining the status of Illinois aquatic life designated uses and determining the causes (agents) and sources (origins) of impairments. These results have thus far been reported in a series of periodic biological and water quality reports for each watershed group (see Appendix A). Watershed monitoring has also been supported by and the Lower Des Plaines Watershed Group (LDWG) since 2018 and it will eventually be incorporated in the IPS. The combined effort now includes the majority of the upper and lower Des Plaines River Basin including major tributary

[2] Federal Water Pollution Control Act, Public Law 92-500 (33 U.S.C. 1251 et seq.) as amended via PL 107-303, Nov. 2002.

watersheds such as Salt Creek, the DuPage River, Hickory Creek, and smaller direct tributary subwatersheds across parts or all of Cook, DuPage, Lake, and Will Counties in Northeastern Illinois. These reports contain detailed findings about the subject watersheds with an emphasis on aquatic life use status, causes associated with biological impairments, descriptions of the stressor causes and sources, and conclusions of interest to each watershed group and their stakeholders. The derivation of causes has evolved over time and has only since 2020 utilized the stressor thresholds developed as part of the NE Illinois IPS framework. Causes are now listed as exceeding the Very Poor, Poor, and Fair IPS thresholds and also with the Restorability factor reported for impaired sites and the Threat and Susceptibility factors for attaining sites. This will provide the watershed groups with not only the causes associated with an impairment, but also the threats for attaining sites. By providing the severity of a cause coupled with the Restorability ranking a watershed group will be able to prioritize their responses to each. Another recently added analysis to some of the watershed bioassessment reports is a nutrient effects assessment that was first employed in 2017 and which, like all other new tools, has been refined as it is used. The baseline data "ingredients" of this approach are being more fully developed across the NE Illinois IPS study area (MBI 2023) for eventual inclusion in the IPS itself as a more routinely available assessment and management toolset.

## The Integrated Prioritization System (IPS)

The IPS is a framework that merges high resolution monitoring data and assessment results with water quality management goals and objectives in order to better guide decision-making at regional and local watershed scales. The IPS houses analyses of complex environmental data about biological indicators and the effect that chemical and physical variables have on the measured and potential condition of the biota and water quality at the site, stream or river reach, and watershed (HUC12) scales. Regionally scoped data of sufficient resolution is used to determine the extent and severity of stream and river impairments and for developing stressor thresholds. IPS development works best where systematic monitoring and assessment has been conducted at the local watershed scale of resolution such that paired biological, chemical, physical, and land use data is available to develop regionally relevant stressor relationships. In places where existing data is spatially insufficient a systematic collection of data to fill gaps was undertaken as a first step towards IPS development.

Making quantitative indicators and tools available to guide and support restoration and protection efforts undertaken by state and local government agencies, local watershed groups, and their respective stakeholders is the major focus of an IPS framework. As such, an IPS is purposed to provide supporting data and information to restore impaired streams and rivers and also to protect high quality sites, reaches, and watersheds from further degradation. The IPS generates a Restorability ranking for impaired sites, reaches, and watersheds and relates it to the primary limiting factors associated with impaired biota. This can be used to prioritize where restoration actions are likely to be the most successful and support designing the most effective restoration actions. For high quality sites that meet or surpass conditions considered to be in attainment, the IPS produces a Susceptibility and Threat ranking that can be used to develop protective actions for streams and their watersheds aimed at minimizing or eliminating the impact of existing or new stressors. Tutorials about water quality basics, WQS, underlying concepts, and methodologies for collecting data and analyzing it to produce the various thresholds and tools that are an integral part of an IPS framework are also featured. The IPS frameworks that MBI have previously developed (MBI 2010, 2015) have been and are being used to identify priority stressors at a detailed spatial scale and to prioritize assessed stream and river reaches for restoration or protective actions. Pre- and post-project monitoring is used to establish the baseline, clarify stress/response relationships, evaluate and predict impacts, and assess the effectiveness of restoration actions. This data is also used to refine the design of future actions based on the improved understanding gained by periodically evaluating the relationship between stressors, some of which do not have a WQS, and biological assemblage responses. The outputs provided by the IPS can be used for an array of watershed management applications and programs, regulatory and non-regulatory alike.

An IPS framework is especially useful when:

1. The jurisdictional setting includes multiple watersheds, river mainstems, and a complex mosaic of pollution sources and other chemical, physical, and stressors;
2. Widespread impairment has been documented in a regional setting that results in the need for large numbers of abatement projects being identified and prioritized; and,
3. Pollution abatement project needs seemingly outstrip the availability of logistical and financial resources to accomplish such within a specified time frame.

An IPS framework, if properly developed and used, can aid in deciding about priorities for immediate vs. longer term projects based on a detailed assessment of the restorability of impaired watersheds, reaches, and sites to meeting their WQS. An IPS also includes an assessment of the susceptibility and threats to waters that are in attainment, thus including the protection of designated uses along with their restoration as an operational focus.

***The IPS as a Model***

The IPS has been referred to as a “model” in a descriptive sense. But is it really a model in the way most think about a tool that has predictive capabilities to support planning and permitting? Adhering to the definition of a scientific model as a physical and/or mathematical and/or conceptual representation of a system of ideas, events or processes the IPS qualifies as a model. Models are used to identify and understand patterns in the natural world by drawing on existing scientific knowledge to offer explanations that enable patterns to be predicted. Such models need to be consistent with observations, inferences, and current explanations about patterns in aquatic ecosystems. The most useful scientific models will possess:

- Explanatory power - a model that contributes little or nothing to explanations of complex processes is of little value;
- Predictive power - the testing of predictions derived from a model is fundamental in establishing the robustness of the model;

- Consistency across contexts - the model of stream quality is the same between different streams;
- Consistency with other scientific models - the model includes relationships and variables that are in common with other models.

A unique quality about the IPS as a model is its dependency on direct observations made about a complex array of biological, chemical, and physical ecosystem attributes at the appropriate spatial scales as opposed to employing assumptions about a handful of chemical variables that simulate assumed outcomes.

Monitoring and assessment is conducted as the first step of IPS development by identifying the most limiting stressors, resolving WQS attainability issues *ahead of* determining the extent and severity of WQS impairments, and delineating associated causes and sources. This produces an informative database that can be queried at the watershed, reach, and site-specific scales by various users who are focused on specific water quality management issues. The IPS produces rankings of restorability, susceptibility, and threat each of which can be used to identify both restorative and protective actions that have the highest return on investment at the watershed, reach, and site-specific scales. As a result, an IPS can assist in setting and responding to required regulatory actions while cost-effectively improving conditions for aquatic life and the attainment of WQS.

***IPS Precedents***

Precedents for developing an IPS include the original prioritization framework developed by Ohio EPA for vetting applications to the Water Resource Restoration Sponsor Program (WRRSP) for habitat restoration proposals, the original Project Identification and Prioritization System of the DuPage River Salt Creek Workgroup[3] (DRSCW; Miltner et al. 2010) in DuPage and Cook Counties, Illinois, and the development of an IPS System for addressing CSO and stormwater issues for the Metropolitan Sewer District of Greater Cincinnati (MSDGC; MBI 2015). A key component that the NE Illinois IPS shares with each of these programs is the explicit goal to protect and restore aquatic life uses and to ensure that such efforts address the limiting factors identified by high resolution watershed monitoring and assessment – the Ohio EPA WRRSP, the first DRSCW IPS, and the MSDGC IPS are each informed by detailed monitoring and assessment information on par with that supported by the five major watershed groups - DRSCW, LDRWC, NBWW, LDWG, and DRWW. The IPS systematically focuses on actions that are designed to address the factors that have been documented by monitoring and assessment as limiting the attainment of aquatic life goals. The NE Illinois IPS tool offers some technical advances based on the lessons learned by the Ohio EPA, DRSCW, and MSDGC efforts in using their respective IPS frameworks.

[3] The goal of the DRSCW is to develop an: "active biological stressor prioritization system to support a quantitative decision-making process for developing restoration options for impaired reaches of streams and rivers in the DuPage and Salt Creek watersheds. The basis for this system is the recent monitoring and assessment results and GIS-based environmental infrastructure information that was developed for these watersheds in 2006-7. The approach included a systematic process that provides reach level information on the most limiting stressors to biological attainment and a rating of the restorability of impaired reaches based on the information and processes contained in the five major factors that determine the integrity of aquatic ecosystems (Karr et al. 1986) which, in turn, are linked to the sources of these stressors." (Miltner et al. 2010).

Given the uncertainties about identifying and resolving impairments in a complex, stressor rich urban setting (Walsh et al. 2005), IPS outputs will be essential for sorting through a mosaic of overlapping stressors some of which are not easy or perhaps even feasible to control. The NE Illinois IPS is underpinned by the identification of the agents of impairment and estimates of the likelihood of restoration. At the same time the NE Illinois IPS can be useful for watershed management and planning purposes by using the susceptibility and threat rankings for protecting rivers and streams that already meet their WQS, which in the DRSCW, NBWW, LDPWC, LDWG, and DRWW areas are few and mostly lie at the boundaries of these watersheds.

The NE Illinois IPS is accessed by the IPS Dashboard which utilizes Microsoft Power BI allowing users to explore various data about NE Illinois streams and rivers that have been ranked by measures of aquatic life *Restorability* for impaired waters and *Susceptibility* and *Threat* for waters meeting the General Use and those with higher quality (Excellent) conditions. It provides ready access to both recent (2006-21) and historical data (pre-2006) with the capability to integrate environmental information about sites, reaches, and watersheds as part of the development of restoration and protection projects and strategies. The NE Illinois IPS also includes information about overlapping stressors such as stormwater, habitat alterations, and legacy (sediment) pollution hence it can be useful for managing those sources throughout NE Illinois. A User Manual (MBI 2020) serves as a guide for navigating and using the IPS Power BI Dashboard that is to be continuously updated as new data becomes available.

## BACKGROUND

The Clean Water Act (CWA) ultimately requires a broad focus on the restoration and protection of aquatic life uses by considering all causes and sources of impairment. The instream data used to develop the IPS is intended by design to guide and support a wide range of actions that have the restoration and protection of aquatic life uses as their principal goal. In addition, the data inherently include attributes and values that are needed to build public support for watershed restoration and protection efforts. The NE Illinois IPS is focused on the aquatic life use goals of the Clean Water Act and Illinois WQS and the causal agents (e.g., pollutants and pollution impacts such as sedimentation, flow alteration, and habitat loss) that influence when and if these goals are ultimately attained. The IPS and its components should prove useful in the development and implementation of watershed action plans (Figure 1) as it provides much of the required information in an organized manner.

While vitally important to the success of water quality management, the collection of monitoring data is not an end in itself, but rather an essential means to an end. Data is made more useful when it is converted to information that can support decision-making about the protection and restoration of streams and rivers via active adaptive management. To accomplish this, complex chemical, physical, and biological data needs to be converted into more easily understood indicators that allow users to graphically visualize a gradient of quality. Ideally, results can be provided on the same scale of quality including biological condition (fish and macroinvertebrate assemblages), water quality, including key chemical and physical parameters (e.g., dissolved oxygen, conductivity, habitat, flow alterations, toxics, etc.,), and major stressors such as land uses (e.g., percent of impervious surface, developed, or forested lands in upstream catchments and in riparian areas), nonpoint sources (e.g., urban runoff), and point sources (CSOs, SSOs, WWTPs). The results of the annual watershed assessments conducted from 2007 to the present were first compared to the Illinois biological (fish and macroinvertebrate) benchmarks and chemical water quality criteria to determine status and identify the causes and sources associated with impairments. The IPS further organizes these results in relation to the restorability of impaired sites and reaches and also by the level of threat and susceptibility to attaining sites by current levels of stressors. Within the IPS, the results can be plotted or mapped in relation to priorities developed by the watershed groups and other stakeholders that can take into account social (e.g., local citizen interest or plans, adjacent parkland, or recreational areas), economic (e.g., cost estimates, restoration costs), or administrative factors (e.g., NPDES schedules, stormwater permits, etc.). A glossary of terms and list of acronyms are included to help translate the jargon commonly used by CWA-based watershed programs.

As illustrated by Figure 1, the NE Illinois IPS is also designed to deliver and visualize the functions of a Watershed Action Plan which includes defining the watershed, assessing the quality of the receiving waters, identifying the key stressors and their sources, identifying high quality aquatic resources, setting goals or thresholds for key stressors, setting priorities, measuring progress, and engaging the public. The integration of rotating watershed monitoring and the IPS more fully accomplishes each of these tasks. This approach also incorporates an innovative "viewpoint" that focuses on the receiving streams compared to common regulatory approaches that focus primarily on sources and water quality at the "end-of-pipe", by assuming that controls based on loading and/or volume reductions will meet ambient water quality goals (i.e., WQS) without necessarily validating that assumption with instream monitoring and assessment. The same can be said for watershed planning that focuses on GIS data and source areas as the principal "commodity" at issue. The NE Illinois IPS views ambient water quality directly by measuring it instream, accounting for the attainment and attainability of WQS, and then relating it back to all sources present, hence it looks from the receiving water back to the source(s). In order to be successful key elements of

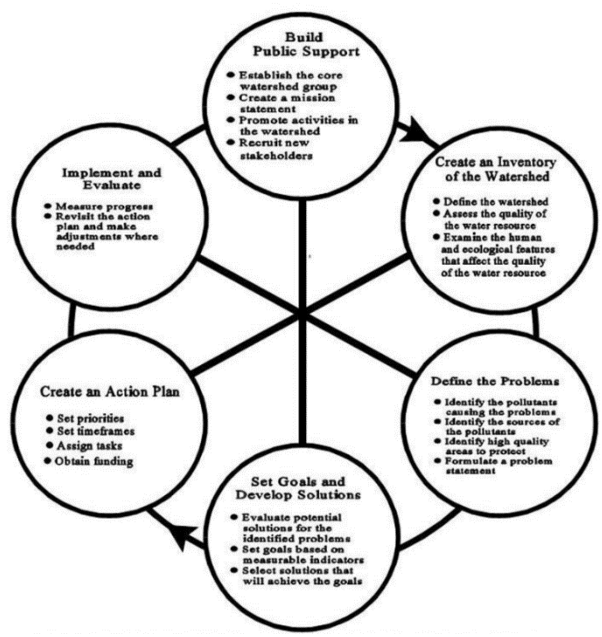



***Figure 1.*** *The IPS directly fulfills the data driven functions of a Watershed Action Plan (WAP) such as creating a watershed inventory, identification of impairments, prioritization, measurement of effectiveness, and engaging the public.*

both approaches are needed, but require appropriate integration in order to be representative, accurate, and cost effective.

## An Adequate Watershed Monitoring Program

The question about what constitutes an *adequate* watershed monitoring and assessment program was articulated in general by the ITFM (1992, 1995) and more specifically by Yoder (1998). Adequate monitoring and assessment were seen as key to resolving the deficiencies and inequities within and between state Clean Water Act (CWA) programs and answering questions about the reliability of 303(d) listings, nonpoint source management, and water quality standards (WQS) that were prevalent in the 1980s and 1990s. *Important Concepts and Elements of an Adequate State Watershed Monitoring and Assessment Program* (Yoder 1998) outlined the important elements and concepts of adequate watershed monitoring and assessment. It relied principally on the results of the ITFM process, U.S. EPA environmental indicator initiatives of the late 1980s and early 1990s (U.S. EPA 1995 a,b), and state agency experiences in operating systematic and adequately funded programs over a period of 25 years (Yoder and Barbour 2009). Table 1 provides an overview of surface water indicators and how each can be related to broad environmental and natural resource management objectives. The data for some of these indicators may be accessed from multiple sources during the analysis and reporting phases of a monitoring and assessment process.

### *Choosing Indicators and Parameters*

Multiples of different types of measurements comprise an important part of the adequate watershed monitoring and assessment approach consisting of core and supplemental indicators and parameters (Figure 2). The core parameters form the essential basis of the design and are collected at all sites and independent of assessment questions because they represent the baseline attributes of an aquatic ecosystem. The role of biological indicators as the direct measures of ecosystem condition and response supported by chemical and physical parameters as indicators of stress and exposure is fundamentally important. These comprise the core indicators of an adequate monitoring and assessment approach and they are used directly to answer fundamental assessment questions and tasks such as overall ecosystem status, water quality standards compliance, use attainability analyses, delineating causes/sources of threats and impairments, and baseline CWA reporting (305b) and listing (303d). Supplemental parameters (Figure 2) consist of chemical, physical, and bacterial indicators of stress and exposure and are added to the core indicators in accordance with the environmental setting and as the assessment questions increase in diversity, density, and complexity.

## The Five Factors of Aquatic Resource Integrity

Taken together the structure of the indicators and parameters reflects the five factors that comprise the integrity of an aquatic resource (Karr et al. 1986; Figure 3). These five factors include:

**Energy source:** changes in the food web, including nutrients, organic material inputs, seasonal cycles, primary and secondary production, sunlight.

***Table 1.*** *Summary matrix of recommended environmental indicators for meeting management objectives for status and trends of surface waters (a bold* ***X*** *is recommended as a primary indicator after ITFM 1992, a lower case x is secondary; additional recommended indicators are designated by a √). The corresponding EPA indicator hierarchy level (U.S. EPA 1995a,b) is listed for each.*

| | **Categories of Management Objectives** | | | | | |
|---|---|---|---|---|---|---|
| | **Human Health** | | | **Ecological Health** | **Economic Concerns** | |
| **Indicator Groups** | **Consumption of Fish/ Shellfish** | **Public Water Supply** | **Recreation (swimming, boating, fishing)** | **Aquatic & Semi-aquatic life** | **Industry/ Energy/ Transportation** | **Agriculture/ Forestry** |
| **<u>Biological Response Indicators (Level 6)</u>** | | | | | | |
| Macroinvertebrates | | x | **X** | **X** | √ | √ |
| Fish | **X** | x | **X** | **X** | √ | √ |
| Semi-aquatic animals | x | | **X** | **X** | x | x |
| Pathogens | **X** | x | **X** | | **X** | |
| Phytoplankton | **X** | **X** | x | **X** | **X** | |
| Periphyton | | | | **X** | | |
| Aquatic Plants | | x | **X** | x | x | x |
| Zooplankton | | x | x | x | | x |
| **<u>Chemical Exposure Indicators (Levels 4&5)</u>** | | | | | | |
| Water chemistry | x | **X** | **X** | **X** | x | **X** |
| Odor/Taste | **X** | **X** | **X** | | | X |
| Sediment Chemistry | **X** | **X** | **X** | **X** | **X** | **X** |
| Tissue Chemistry | **X** | x | | x | x | |
| Biochemical Markers | √ | √ | √ | √ | | √ |
| **<u>Physical Habitat/Hydrological Indicators (Levels 3&4)</u>** | | | | | | |
| Hydrological Measures | x | **X** | x | **X** | **X** | **X** |
| Temperature | x | x | **X** | **X** | **X** | **X** |
| Geomorphology | x | x | **X** | **X** | **X** | **X** |
| Riparian/Shoreline | x | **X** | √ | **X** | x | **X** |
| Habitat Quality | √ | √ | √ | √ | √ | √ |
| **<u>Watershed Scale Stressor Indicators (Levels 3,4,&5)</u>** | | | | | | |
| Land Use Patterns | x | **X** | **X** | **X** | x | **X** |
| Human Alterations | x | **X** | **X** | **X** | x | √ |
| Watershed Imperv. | √ | √ | √ | √ | √ | √ |
| **<u>Pollutant Loadings Indicators (Level 3)</u>** | | | | | | |
| Point Source Loads | √ | √ | √ | √ | √ | √ |
| Nonpoint Loadings | √ | √ | √ | √ | √ | √ |
| Spills/Other Releases | √ | √ | √ | √ | √ | √ |

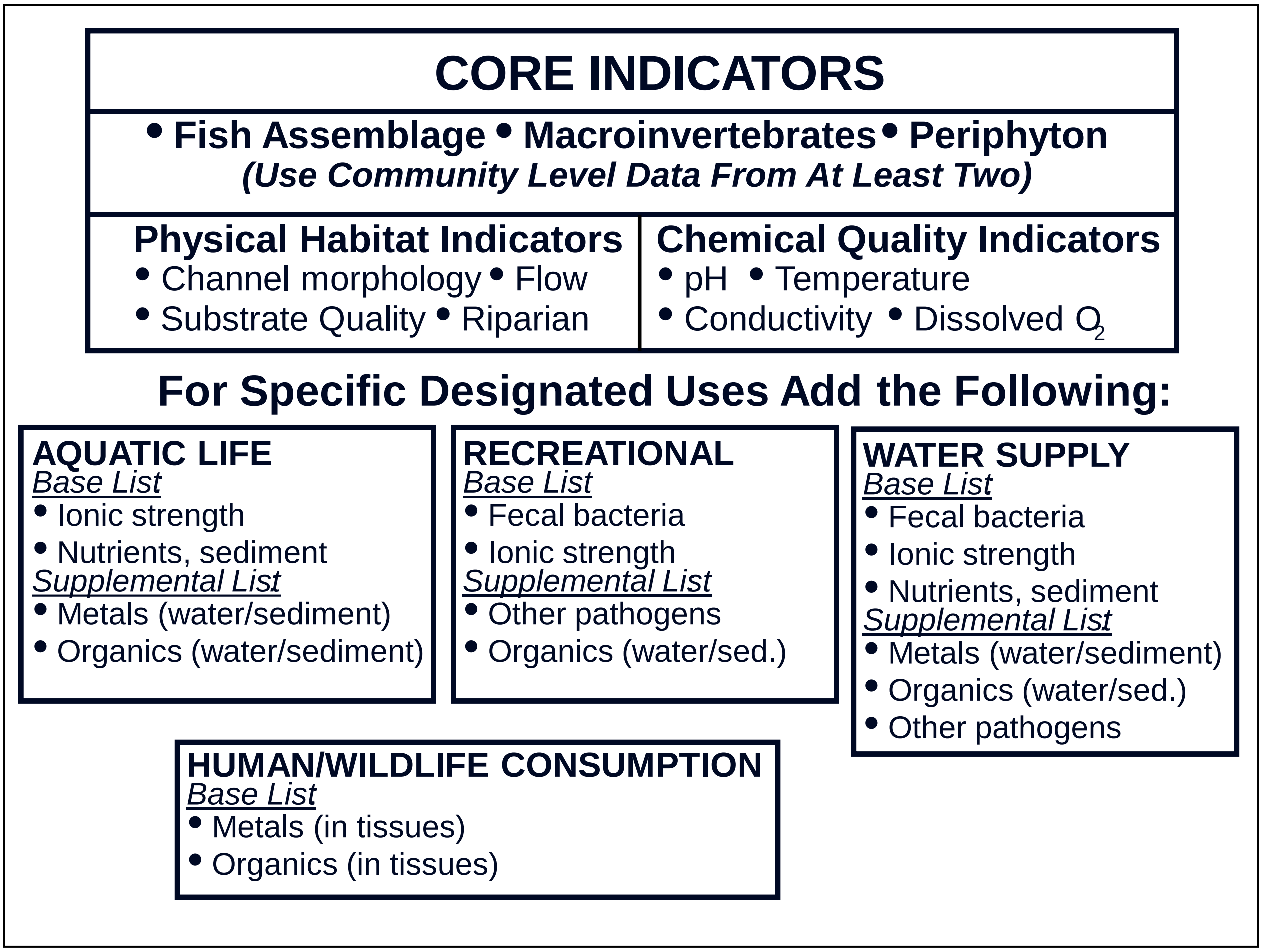



***Figure 2.*** *Core and supplemental indicators and parameters arrayed by designated uses. This structure illustrates how specific indicators and parameters are chosen for data collection and analysis under an adequate watershed monitoring and assessment approach (after Yoder 1998).*

**Chemical variables:** changes in chemical water quality, including D.O., pH, turbidity, hardness, alkalinity, ionic strength, nutrients, organics, toxic substances, temperature, sediment and their modes of action (e.g., solubility, adsorption, etc.).

**Flow regime:** modification of flows, including precipitation, seasonal patterns, runoff, velocity, groundwater, flow extremes.

**Habitat structure:** alteration of physical habitat, including bank stability, current, gradient, instream cover, vegetative canopy, substrate, current, sinuosity, width, depth, pool-to-riffle ratios, riparian vegetation, sedimentation, channel morphology.

**Biotic factors:** changes in biotic interactions, including alien taxa, feeding, reproduction, predation, overharvest by sport, commercial, and subsistence fishers, diseases, parasitism, and competition.

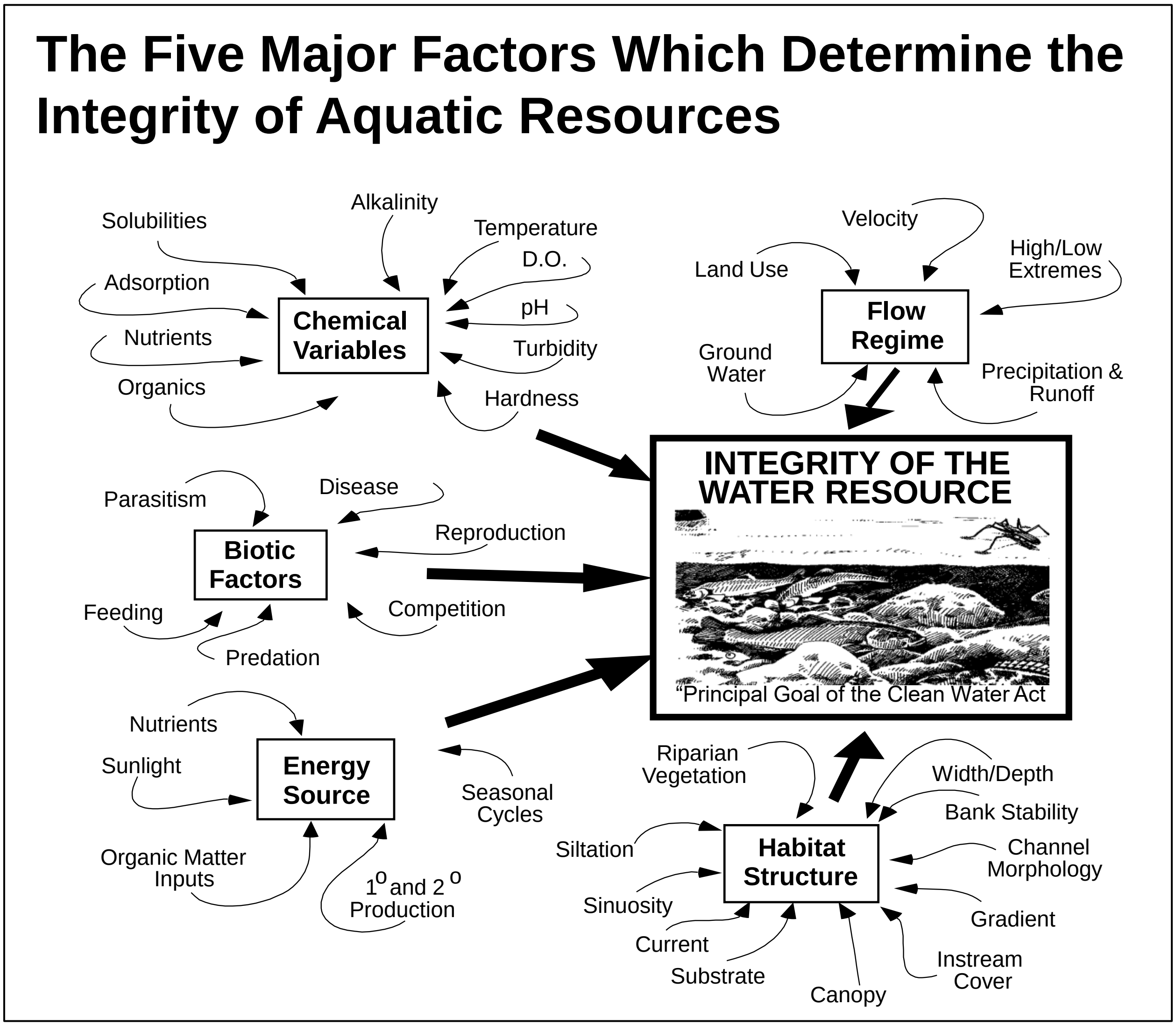



***Figure 3.*** *The five factors that comprise and determine the integrity of an aquatic resource (after Karr et al. 1986).*

When stressors influence or impact one or more of these factors, or their interactions, the aquatic biota responds in a predictable manner as depicted in Figure 4 which also serves as an explicit model of causation (Karr and Yoder 2004). The severity and extent of the biological response to these impacts are ultimately what is important, not the mere presence of an impact itself. It is the understanding of these interactions that is used to guide the selection of indicators and parameters for comprehensive monitoring programs (Karr 1991; Yoder 1998).

This framework emphasizes cost-effectiveness by carefully allocating sampling resources and by scaling the intensity and complexity of the monitoring in accordance with the complexity of the local setting and the management issues that need to be addressed or exposed. It also allows for more flexible management responses that are attenuated by the information revealed about the environmental complexity of the setting, the inherent quality of the aquatic resource, and the types of pollution problems that are encountered. Effective implementation is enhanced through experience and knowledge gained by conducting monitoring and assessment for many years and over a wide geographical area.

Adequate monitoring employs a stepwise approach to the selection and use of chemical, physical, and biological indicators and measures that are currently available. The decision(s) about which indicators and parameters to use are based on the type of aquatic resource being assessed (i.e., headwater stream, wadeable stream, non-wadeable large river, lake or reservoir, wetland, etc.), the environmental complexity of the setting (including the consideration of all potential stressors), and the water quality management objectives and purposes that are at stake. For example, in a small, headwater stream with only one or two potential stressors, the two biological assemblage groups are accompanied by a qualitative habitat assessment, and a comparatively limited chemical water quality analysis for field, demand, and nutrient

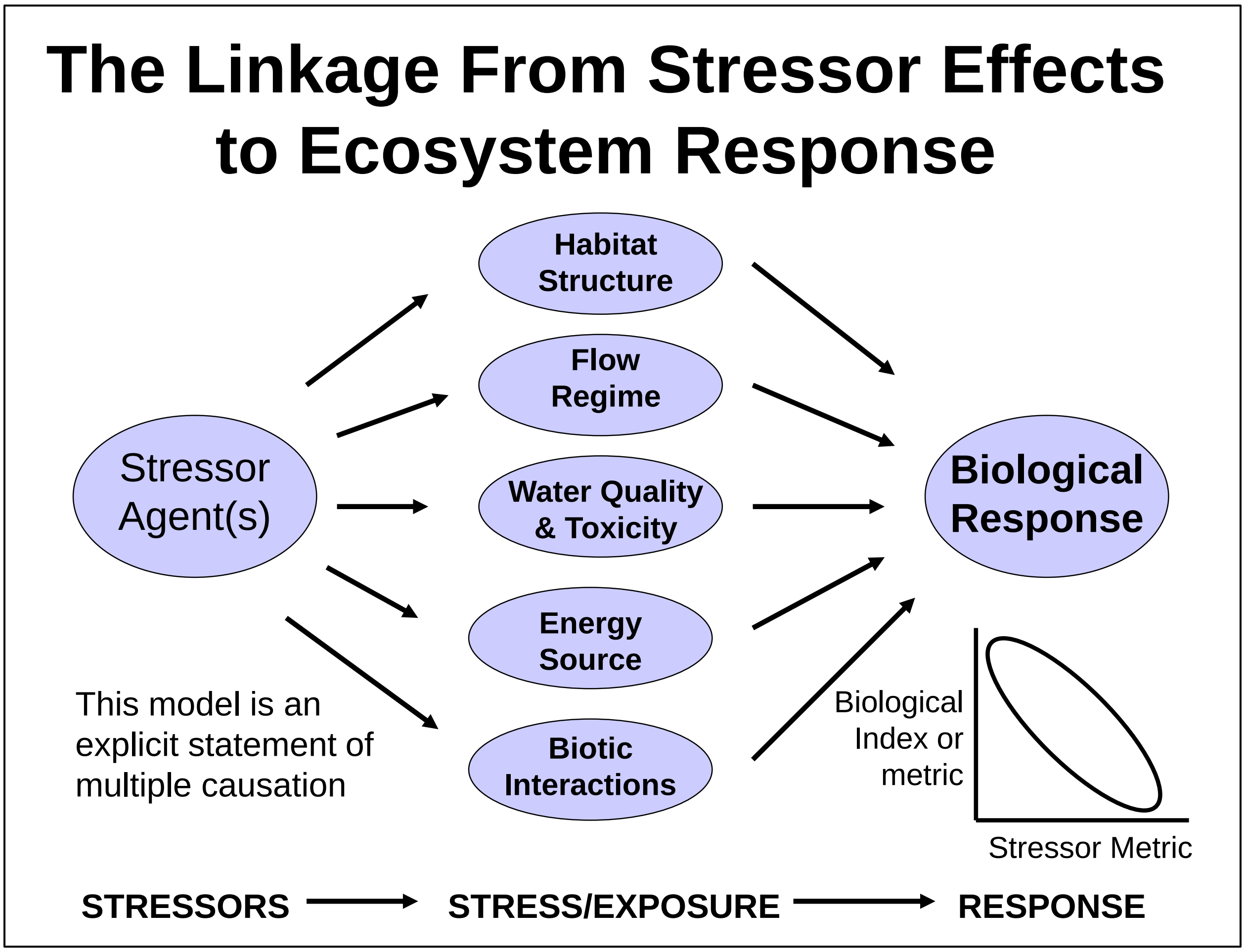



***Figure 4.*** *Linkages between stressors (or drivers of ecosystem change) through the five major factors of water resource integrity (as altered by stressors) to the biological responses produced by the interactions. The biological response is the endpoint of primary interest and is the focus of water quality management. This model illustrates the multiple causes of water resource changes associated with human activities. The insert illustrates the relationship between stressor dose and the gradient of biological response that signals a good biological metric (after Karr and Yoder 2004).*

parameters. Field sampling can be completed with one visit for biology and habitat and 2-3 samples for chemical/physical parameters within a seasonal index period. Multiples of these sites can be sampled in a field day. In more complex watershed settings with multiple management issues, multiple and complex stressors, and the potential for the discovery of undocumented stressors, the cumulative sampling requirements become more intensive with two visits for biology and habitat and more frequent (up to weekly) sampling for a more complex array of chemical analyses in water and sediment including heavy metals and organics. Emerging concerns about the effects of nutrient enrichment have resulted in adding continuous monitoring for temperature, pH, and dissolved oxygen (D.O.), and analyses of chlorophyll α on the substrates (benthic) and in the water column (sestonic). A systematic sampling effort spans a summer-fall index period, usually mid-June through mid-October. Data analysis and reporting culminate in the production of a comprehensive assessment several months after the sampling is completed. This ensures that samples are processed, data is managed and verified, and a careful analysis of multiple indicators and assignments of causes and sources can be performed in accordance with sound practice and procedures.

**Spatial Monitoring Design**

Using an information-effective spatial monitoring design is but one of the critical steps in the process of developing an adequate watershed monitoring program. The DRSCW specifically requested a monitoring design which reflects the concepts outlined for an adequate watershed monitoring and assessment program at the outset of their program in 2006. Initially memorialized in an overall plan (MBI 2006) it has served as the template for the watershed based monitoring and assessment that has been conducted across five different groups in Northeastern Illinois since 2006. It was developed under the following general principles and concepts by:

1. Employing the principles of adequate monitoring (ITFM 1992, 1995; Yoder 1998) in selecting indicators and parameters and casting their role as indicators of stress, exposure, and response;

2. Employing an intensive pollution survey design that evaluates pollution from all sources and which is in keeping with its definition in the CWA[4]. It is an observational approach that requires more sites than a condition survey which relies on the extrapolation of data from a handful of sampled sites to many more unsampled sites and reaches.

3. Deriving regionally-based stressor thresholds using the database generated by the paired collection of biological, habitat, and chemical/physical data and subsequent analyses utilizing attributes of the two primary biological assemblages, fish and macroinvertebrates; and,

4. Providing a spatially comprehensive regional dataset to develop an Integrated

[4] The CWA defines *pollution* as the human-induced alteration of waters caused by pollutants *and* non-pollutant agents, such as flow alteration, physical habitat alteration, and introductions of alien taxa [CWA section 502(19)].

Prioritization System (IPS) and affiliated “tool set” for more effectively managing river and stream quality and for setting priorities for restoration and protection.

The spatial monitoring design employs a combined geometric (stratified-random) and targeted-intensive pollution survey. This design supports the determination of the status of aquatic life and recreational use attainment at the same scale at which pollution sources are being managed and regulated within the NE Illinois watersheds. Given that there are hundreds of point sources, thousands of stormwater structures, varying degrees of urban and suburban development, sites that contain legacy pollutants, and a gradient of habitat alterations, the intensive pollution survey design is needed to capture and characterize the numerous and overlapping pollution gradients that result from these sources. As such, the monitoring design adhered to the principles of adequate monitoring (ITFM 1995; Yoder 1998). Doing so assured that the resulting watershed assessments could be used to support the development of cost-effective responses to the existing array of pollution sources and provide information that also supports management responses to other sources and planning for future development.

***Intensive Pollution Surveys vs. Condition Surveys***

The intensive pollution survey design exemplified by the DRSCW baseline monitoring design is fundamentally different from the condition surveys that are commonly employed by most state and federal agencies in the U.S. at present. Some of the key characteristics, outputs, and outcomes of each are described in Table 2 to offer a perspective about what each offers and some of the inherent assumptions that each may impose on the uses of the data and results to support water quality management programs and their functions. The differences are necessarily portrayed as a binary comparison, but are in reality, a gradient between each in terms of their capacity to deliver support to water quality management programs.

A combined Intensive Pollution Survey and Geometric Site Selection design (hereafter Intensive Pollution Survey) is intended to provide data that is spatially sufficient to assess impairments and their causes and sources at the site, reach, and watershed (12 digit Hydrologic Unit Code or HUC12) scale. This design is based in part on the pioneering concepts about pollution gradients described by Bartsch (1948) and Doudoroff and Warren (1951) and from a collection of papers compiled by Keup et al. (1967) to facilitate the detection and quantification of degradation and recovery from pollution influences along a river or stream reach (i.e., pollution impact reaches). Multiples of sampling sites are located upstream from major sources of disturbance, in areas of immediate impact and potentially acute effects, through reaches of increasing and lessening degradation, and reaches of recovery. It is not a point source only focused approach, but rather a pollution focused approach the latter of which can emanate from either point or nonpoint sources. The more complex the array of stressors and their spatial extent the more that pollution gradients will overlap and intermix forming a veritable mosaic of stress and response relationships.

***Condition Survey Design***

Condition surveys are intended to provide a broad assessment of aquatic resource condition at a regional, statewide, or national scale by sampling a subset of representative sites that provides a statistical basis for extrapolating the results to all unsampled sites. They are regarded as being more cost-effective than intensive surveys because of the lower per capita field effort and there are examples where relationships with stressors have been extracted. However, few have questioned how the paucity of data along pollution gradients at the site, reach, and watershed scales affects the ability of condition surveys to detect and quantify stressors that are apparent only at the site, reach, or small watershed (e.g., HIC12) scales. Furthermore, stressors that act at multiple scales may escape complete detection and adequate characterization. In addition, river and stream networks have inherent properties such as dendritic branching, directed flow, and abrupt changes in physical, chemical and biological attributes at tributary junctions (Peterson and Ver Hof 2014; Ver Hof et al. 2014) *and with changes exerted by non-randomly positioned point and nonpoint sources of pollution.* Some of the more commonly employed condition survey designs ignore these properties, which increases the chances of biased results and poor scientific inference. Finally, most restoration and management of impaired waters takes place at the reach and site scales, hence spatially sparse assessment designs are insufficient to identify and prioritize such efforts. Simply extrapolating (i.e., kriging[5]) widely spaced sites across a landscape ignores these inherent properties of rivers and streams and their watershed networks, especially in heterogeneous urban watersheds. The value of intensive pollution surveys have largely been discounted by designs that emphasize the assessment of condition in fulfillment of CWA 305[b] reporting and 303[d] listing objectives. The latter simply provides inadequate spatial coverage and incomplete stressor gradients to organizations seeking to address pollution issues at the site, reach, and small watershed scales. Table 2 offers a qualitative and quantitative accounting of the differences between condition and pollution surveys.

***Pollution Survey Design***

An important goal of a pollution survey is to determine the relative effects of multiple sources along spatial and stressor gradients. Large mainstem rivers and streams are treated as distinct units to understand how changes take place along a longitudinal pollution continuum with respect to both natural and anthropogenic influences which is where the majority of pollution survey sites are located. Tributaries are accounted for via the geometric allocation of sites by panels of drainage area. It yields a detailed assessment of impairments, their extent and severity, specific indicator responses in stream and river reaches along pollution gradients, and assessing temporal changes when applied in the context of a sequential rotation of river basin monitoring units.

The intensive pollution survey design is employed within watersheds that correspond to a HUC10-12 scale in order to fulfill multiple water quality management objectives in addition to a more conventional focus on general status assessment. This is the spatial scale that is representative of how CWA management programs are inherently applied – at the site, reach,

[5] Kriging refers to a group of geostatistical interpolation methods in which the value at an unobserved location is predicted by a linear combination of the values at surrounding locations, using weights according to a model that takes into account the spatial correlation that provides unbiased estimates with minimum variance.

***Table 2**. Key characteristics of condition-focused and intensive pollution survey monitoring designs in terms of spatial organization, sampling site density, outcomes, CWA program support, stressor identification, and the capacity for detecting and dealing with cumulative effects (from MBI 2019).*

| Key Characteristics | Condition Monitoring | Pollution Monitoring |
|---|---|---|
| **Spatial Organization** | • Probabilistic<br>• Synoptic (non-random)<br>• "Pour point" (HUC8-12) | • Sites, Reaches, Sub-watersheds (HUC12)<br>• Along longitudinal pollution gradients |
| **Sample Site Density** | • >25 mi.$^2$ per site[1]<br>• 10-25 miles per site[1]<br>• 1.5 avg. sites per HUC12[1]<br>• 4.6 avg. sites per HUC10[1] | • 1.5-3.0 mi.$^2$ per site[2]<br>• 1-5 miles per site[2]<br>• 10.4 avg. sites per HUC12[2]<br>• 59.3 avg. sites per HUC10[2] |
| **Outcome(s)** | • Delineate status over wide area (regional, statewide)<br>• First order stressor identification | • Delineate status at the site, reach, and watershed (HUC12) scales.<br>• Delineate pollution gradients.<br>• Quantify severity and extent of reach scale impacts.<br>• Detailed stressor identification.<br>• Use attainability analysis. |
| **CWA Program Support** | • 305[b]/303[d] reporting & listing.<br>• Indirect support for implementation (TMDL, NPDES, etc.) | • 305[b]/303[d] reporting & listing.<br>• Direct support for implementation (TMDL, NPDES, WQS, 404/401, stormwater, planning, BMPs). |
| **Stressor Identification** | • First order determination of stressor relationships (limited by scale). | • Detailed delineation of stressor relationships across watershed strata.<br>• Regional development of stressor thresholds. |
| **Cumulative Effects** | • Too few sites at HUC12 scale to distinguish site- and HUC-specific (cumulative) effects of stressors.<br>• Insufficient sites to reveal pollution gradients and profiles. | • Multiple sites at HUC12 sufficient to distinguish site- and HUC-specific (cumulative) effects of stressors.<br>• Sufficient sites to reveal longitudinal pollution gradients and profiles. |

[1] IEPA/DNR surveys in DRSCW/DRWW watersheds in NE Illinois 2006-18.
[2] DRSCW/DRWW watershed surveys in NE Illinois 2006-18.

and HUC12 watershed scales. In the Midwestern U.S., most HUC10 watersheds drain approximately 150-300 $mi^2$, although this is not invariable. Sites within a watershed of this size are allocated based on a geometric progression of drainage areas starting with the area at the mouth of the HUC10 watershed and working upstream through the mainstem and tributaries to the primary headwaters. Sampling sites are geometrically allocated according to the stratification of available stream and river sizes based on the drainage patterns. A targeted selection of sampling sites that are needed to focus on localized sources such as point source discharge mixing zones and pollution impact reaches, habitat modifications, dams and impoundments, and other potential impacts within a watershed are added to comprise an intensive pollution survey design. This combined design fosters data analyses that can better account for the overlying and variable natural and human caused influences within the streams and rivers of a watershed and collectively between watersheds. It simultaneously supports multiple water quality management needs including the *proportionate* assessment of all streams and rivers, applying a gradient of narrative condition ratings, the development of more complete TMDLs that include the inter-relationships of both pollutant and non-pollutant stressors, and the development of a comprehensive and spatially representative database through time. Other benefits of this design include the application of cost-effective sampling methods on a watershed scale, development of a spatially comprehensive database, and an enhanced ability to include previously unassessed or under-assessed waters. This design has been particularly useful for watersheds that are targeted for TMDL development such that unassessed or under-assessed waters, incomplete or outdated assessments, and outstanding or unforeseen WQS attainability issues can be addressed *prior to* TMDL development as is illustrated in Figure 5.

The principal outputs are based on an interdisciplinary monitoring effort coordinated on a water body specific or watershed scale. Biological, chemical, and physical monitoring and assessment techniques are employed as bioassessments to meet three major objectives:

1. Determine if use designations and/or goals set for or assigned to a given water body are appropriate and attainable;

2. Determine the extent to which use designations (or equivalent classifications) assigned in the State WQS (or policies) are either attained or not attained and with the assignment of causes and sources for the latter; and,

3. Determine if 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 for nonpoint sources.

The data gathered in a bioassessment is processed, evaluated, and synthesized in a comprehensive assessment report that addresses use attainability issues, future monitoring needs, problem discovery, or other actions which may be needed to resolve impairments or threats to designated uses. While the principal focus of a bioassessment is on the status of aquatic life, the status of other uses such as recreation and water supply, as well as human

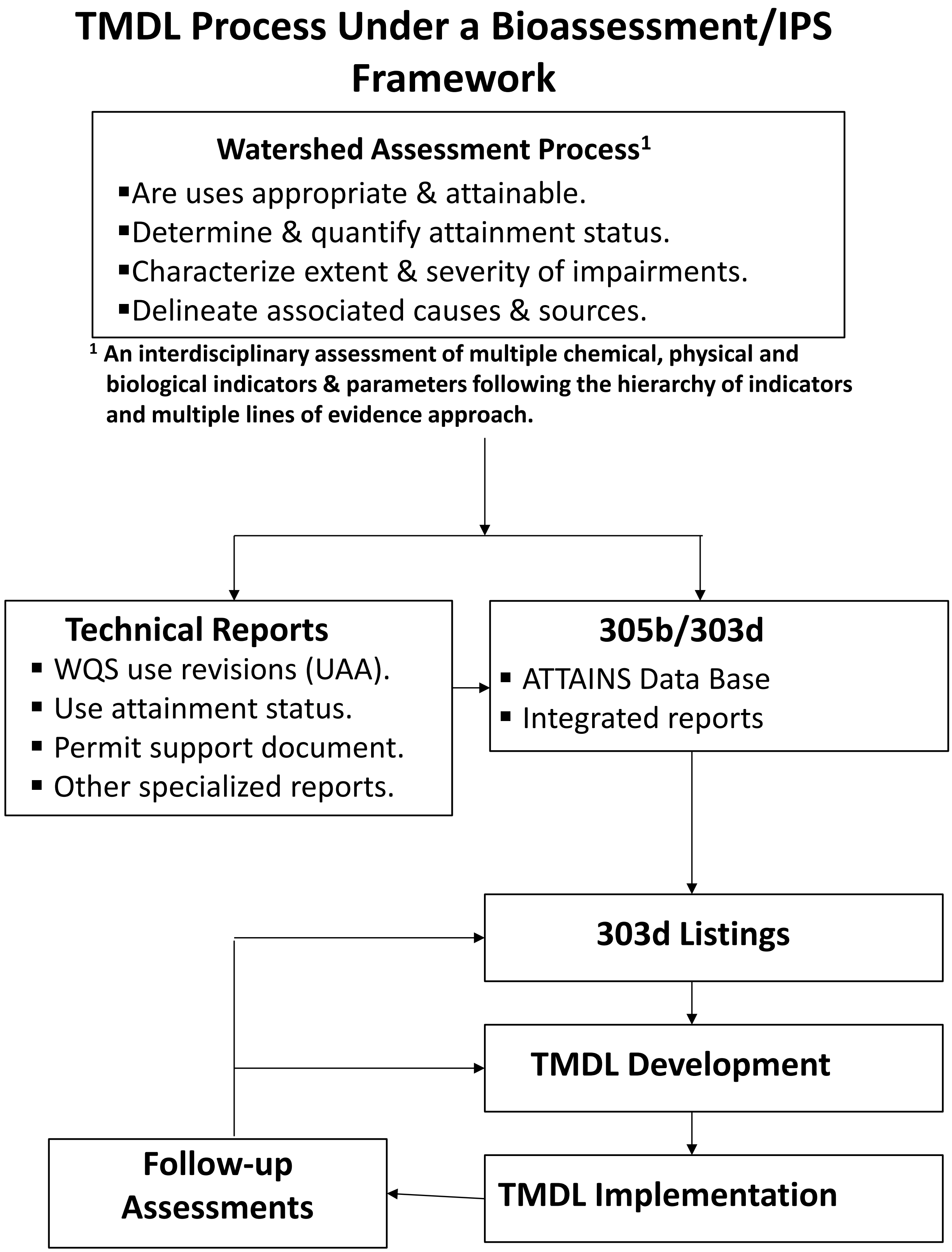



***Figure 5.*** *Key steps in a TMDL development and implementation process under a bioassessment based IPS framework supported by an intensive pollution survey monitoring design.*

health concerns can also be addressed provided the data in support of those uses is collected and analyzed within the monitoring and assessment design.

## Stressor Identification

Once a biological impairment is identified, a next step is to identify the responsible causes (i.e. agents or stressors) and sources (i.e., origin of stressor) for an impairment. Adequacy in stressor analyses is important partly because the cost of point source and stormwater remediation can be high and initial estimates rarely include a careful enough consideration of ecological impacts (Visitacion et al. 2009). MBI employs a lines-of-evidence approach where multiple types of data (e.g., biological responses, water quality criteria or other thresholds, habitat data, land use, etc.,) are used in a "stressor identification" process (SI) to identify associated causes/sources and their relative contributions to an observed impairment. The need for such an approach is well summarized by (Vander Lann et al. 2013):

> *"Cause and effect can rarely be established from single studies (Norris et al. 2012), so a weight-of evidence approach generally is needed to identify the most likely causes of impairment (Suter et al. 2010). Strong inferences regarding the causes of ecological degradation require, at a minimum, observed exposure of biota to a stressor, identification of a plausible causal mechanism (i.e., a causal chain starting with exposure and ending in a biological response), and a consistent and strong association between the hypothesized cause and effect (Norris et al. 2012)."*

Data collected from large scale synoptic sampling[6] programs using robust sampling approaches can be used to develop thresholds and other targets and this data can be used to understand how stressors limit aquatic life under ambient conditions. As restoration efforts remove or alleviate certain stressors over time (e.g., wastewater treatment loadings) or as other stressors increase over time (e.g., residual chloride build-up from road salt), underlying databases will need to be re-examined to determine if new combinations of environmental conditions exist that can provide further insight into causal relationships between stressors and biological response. For example, in the 1980s and 1990s point source pollutant loadings of ammonia and oxygen demanding wastes were reduced via improved wastewater treatment mandated by water quality-based NPDES permit limitations. However, chloride in urban runoff has increased since this time period such that it now poses a realistic threat to aquatic life improvements. The sustained collection of data that is part of the watershed groups rotating watershed approach improves the precision and reliability of predicting changes in environmental stressors over time (i.e., it improves the ability to use statistical controls) and thus the power to distinguish among stressors that may be limiting to aquatic life.

Norton et al. (2009) advocated for using science-based recovery potential screening tools to prioritize restoration of impaired waters. The risk of using a case-by-case or "worst-first"

[6] Synoptic sampling is where many samples are taken during a short time frame (e.g., summer-fall index period) to obtain a spatially comprehensive estimate of conditions in one or more watersheds.

approach to dealing with impaired waters without the systematic consideration of recovery potential can have several undesirable outcomes; 1) more restorable waters may be overlooked resulting in a lost opportunity for having more certain environmental gains; 2) limited restoration resources can be depleted by a relatively few, severely impaired reaches or watersheds that may never recover, thus making it difficult to demonstrate program success; 3) priority-setting without a transparent and consistent basis may be vulnerable to political or legal pressures; and 4) the tools and scientific knowledge about recovery processes are not being fully utilized in restoration decisions that are intended to bring about recovery (Norton et al. 2009).

While uncommon, there have been similarly scoped efforts to develop improved stressor identification and delineation results. The outcomes of six (6) similarly scoped studies that were based on similarly scoped datasets and/or properties in line with those of the NE Illinois IPS were examined (Table 3). While it may seem that this is not an exhaustive compilation of such studies, those that emulate the spatial survey design, scope, parameters analyzed, and major outcomes in terms of using stressor based analyses to measure progress and support large scale restoration planning are rare (Happel and Gallagher 2020). This is likely due a dearth of adequate monitoring and assessment efforts among the states and across the U.S. due to an over-emphasis on national and statewide condition assessments. Of the six studies listed in Table 3, three utilized parts of the Ohio EPA river and stream monitoring database and one (Capmourteres et al. 2018) occurred mostly in the NE Illinois IPS study area based on data from 1972-1976. Stein et al. (2022) utilized a statewide dataset in California based on bioassessment data.

Kroll et al. 2019 cited four major challenges associated with the planning, execution, and monitoring of large-scale restoration programs aimed at improving the ecosystem integrity of streams (and their downstream rivers and estuaries), with a focus on agricultural best management practices and urban stormwater control measures. These challenges are:

1. The lack of holistic planning for implementing and monitoring large-scale restoration projects;
2. Planning that does not include geographic context or considerations of scale;
3. A failure to tie monitoring to specific goals and predicted improvements in ecosystems; and,
4. The limited and parochial approach to monitoring taken by funding agencies.

Thus, the approach first taken by DRSCW and now followed by the other NE Illinois watershed groups certainly addresses these challenges by taking the important first step of supporting an adequate monitoring and assessment program with an important end result being the facilitation of an IPS framework to support restoration planning and assessing its effectiveness.

***Table 3**. A compendium of regional scale studies that have similar characteristics to the NE Illinois IPS framework and for the development of stressor identification, response patterns, and/or causal ranking as a principal outcome.*

| Source/Data Years | Scope/Location/Sites | Spatial M&A Design(s) | Stressor Variables (number examined) | | | | Key Response Variable(s) | Data/Statistical Analyses | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | Chemical | Physical | Land Use | Other | | Stressor Thresholds[a] | Response Patterns[b] | Cause Ranking[c] |
| MBI (this study)<br>2006-2018 | Regional Intra-state<br>Northeastern Illinois<br>n = 640 | Intensive Pollution Surveys | 139 WQ<br>(31[d])<br>144 Sed.<br>(30[d]) | QHEI<br>(17 QHEI attributes[d]) | 36<br>(9[d]) | NA | Fish Species<br>Macroinvertebrate Taxa, fIBI, mIBI | WSVs<br>SSDs | RFM<br>FIT<br>Mapping | R/S/T |
| Bedoya et al. (2005)<br>1996-2000 | Statewide (OH, MN);<br>Regional Intra-state MD)<br>n = 800 | Intensive Pollution Surveys (OH); Basin Surveys (MD, MN) | 21 (OH)<br>9 (MN)<br>8 (MD) | 9 (OH)<br>25 (MN)<br>21 (MD) | 3 (OH)<br>5 (MN)<br>3 (MD) | NA | Fish IBI (all)<br>Macroinvertebrate IBI (OH, MD) | NA | Knn, SOM | NA |
| Zipj et al. (2017)<br>2000-2007 | Statewide (OH)<br>n = 1826 | Intensive Pollution Surveys (OH) | 5+ | 1 | NA | NA | Fish IBI, Species Richness | EVDs | NA | NA |
| Capmourteres et al. (2018)<br>1972-1976 | Regional Intra-state<br>Northeastern Illinois<br>n = 231 | Basin intensive surveys (IL) | 17 | NA | 2 | NA | Fish IBI (old)<br>IBI Metrics (old) | NA | CCA | SEM |
| Kapo and Burton (2006)<br>1990-1996 | Regional Intra-state (OH)<br>Southwestern Ohio<br>n = 300+ | Intensive Pollution Surveys (OH) | 15 | 8 | NA | %Effluent, Modeled (7) | IBI, ICI, Darter Species, Mayfly Taxa | WOE/WLR<br>IAV | GIS (IDW)<br>IPM | NA |
| Whitney et al. (2018)<br>1963, 2017 | Watershed Scale<br>Southwestern Kansas<br>n = 25 | Intensive Watershed Survey (KS) | 6 | 1 | 5 | NA | Site occupancy<br>Relative abundance<br>Species Richness (Fish) | NA | AOD<br>GLMs | NA |
| Stein et al. (2022)<br>1990s-2010s | Statewide (CA)<br>Watershed/Reach Scale<br>n = 1516 | Basin surveys Statewide Probabilistic (CA) | Not provided | CRAM | Stream Cat<br>NLCD | NA | Macroinvertebrate CSCI; Algal ASCI | WLR (land use only) | RFM | R/M/P/RR |

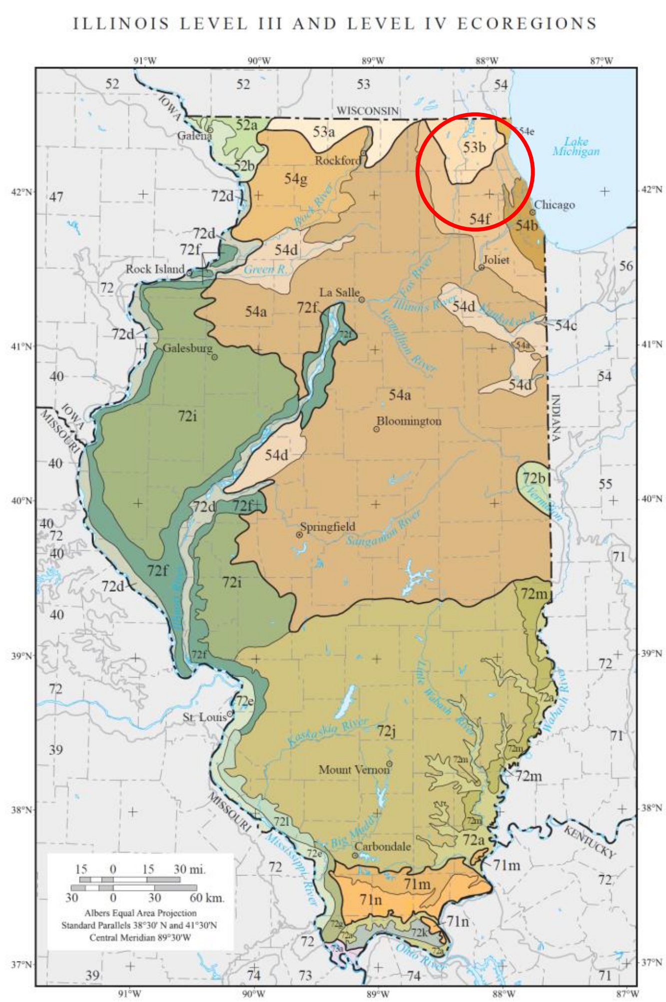
[a] WSV – weighted stressor value; SSD – sensitive species distribution; EVD – ecosystem vulnerability distributions; WOE weight of evidence; WLR – weighted logistic regression; IAV – impairment association values.
[b] RFM – random forest model; FIT – fit function for stressor threshold relationships; Knn – K nearest neighbor; SOM – self-organizing maps; CCA – canonical correspondence analysis; IDW – inverse-distance weighting; IPM – impairment probability models; AOD – analysis of deviance; GLM – generalized linear models.
[c] R/S/T – restorability, susceptibility, threat; SEM – structural equation modeling; R/M/P/RR – restoration, management, protection, risk reduction.
[d] water column parameters for which valid stress:response thresholds could be derived.

# Chapter 2. IPS Concepts and Methods

## THE IPS REGIONAL STUDY AREA

The principal geographic focus of the IPS framework is on the DuPage River, Salt Creek, and Des Plaines River basins and adjacent watersheds to include sites with a wider range of conditions (e.g., including least impacted subwatersheds) to allow the derivation of environmentally meaningful stressor thresholds. Stressor variables across a gradient of quality that spans least- to most-impacted conditions and which have similar natural background conditions is required. Figure 6 shows the location of the general study area in Northeastern (NE) Illinois with respect to Level IV ecoregions (Woods et al. 2006) with the data coming from seven (7)subregions (53a, 53b, 54a, 54b, 54d, 54e and 54f).

One of the lessons learned from the initial IPS developed in 2010 within the DRSCW watersheds was the dominance of fair and poor quality sites and the absence of good and excellent quality sites. This had the undesirable effect of truncating some of the stress:response gradients which were obvious at that time (Miltner et al. 2010). Some external data reflecting better conditions was added to that early version of the IPS, but it did not make the gradient sufficiently complete. As a result, the



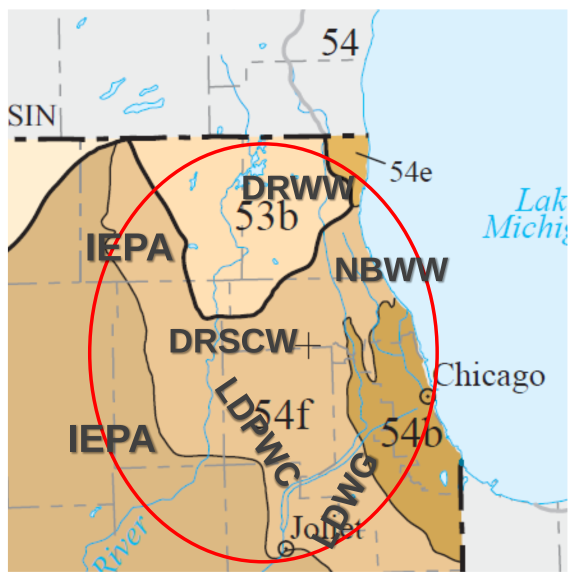




***Figure 6**. The Level III ecoregions and Level IV subregions of Illinois (left) with a zoomed insert into the Northeastern Illinois IPS (NE Illinois IPS) study area with data sources listed by their acronyms (above). Maps are from Woods et al. (2006).*

need to incorporate more data from a wider geographic area was addressed as part of defining the NE Illinois IPS study area.

Biological conditions and water quality across NE Illinois exhibits a wide range of quality from very poor in streams impacted by urban land uses, habitat, flow alterations, and wastewater effluent, to sites with very good to exceptional assemblages located mostly in agriculturally dominated watersheds. The latter was expanded by adding reference sites and historical data collected by IEPA and IDNR in adjoining watersheds. While the immediate focus of the NE Illinois watershed groups is on the elimination of aquatic life impairments, the concepts of susceptibility and threat were also included for protecting the highest quality sites, reaches, and watersheds. This includes sites that currently attain or surpass the Illinois General Use aquatic life benchmarks. By doing this, sites that had fIBI and mIBI scores in the Good and Excellent condition categories consistent with the IEPA narrative ranges were included in sufficient numbers to account for the higher quality portions of the quality gradient for chemical, habitat, land use, and biological indicators.

**Geographic Scale**

Data in the NE Illinois IPS study area was nested in various watershed scales (HUC10 and 12 units) and then by stream and river reach (Illinois Assessment Units – AUIDs) and then by station or site. The IPS study area also included adjacent watersheds to the West and South that are in common to the subregions that encompass the study area (See Figure 6). These data were included to provide a more complete scale for the stressor-response relationships that were used to derive chemical, physical, and land use stressor thresholds applicable to NE Illinois rivers and streams.

The IPS data were analyzed at multiple spatial scales including the HUC10 and HUC12 watershed scales, at the stream or river reach scale, and at the sampling site scale. Scale is important because many of the impacts that limit aquatic life are spatially cumulative (Figure 7) and with pollutants and other stressors acting along pollution continuums from upstream to downstream. Aquatic life can also transition seasonally between different reaches making the connectivity of stream reaches within river basins important. The presence of refuges from stressors or the lack thereof may determine whether a species can

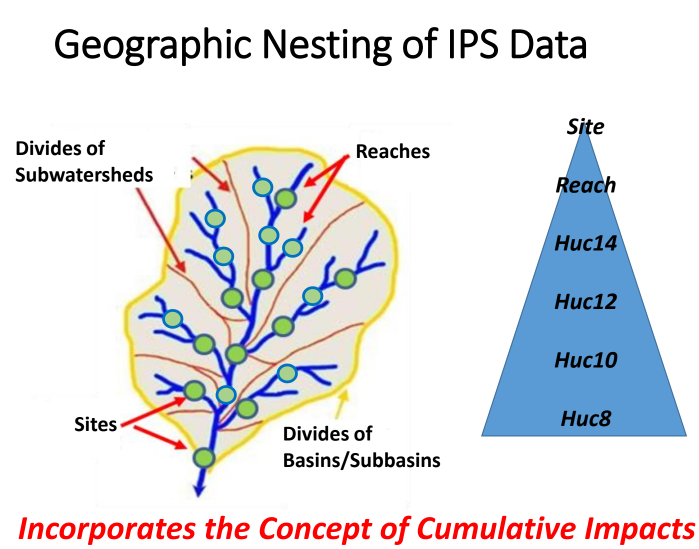



***Figure 7.*** *Watershed boundaries (divides) and stream reaches in relation to sampling sites.*

persist as a viable population in a watershed. A prominent connectivity issue in NE Illinois is the presence of numerous low-head dams that are barriers to fish movement as evidenced by analyses of species occurrence upstream and downstream in several bioassessment reports. An initial attempt to identify all major dams in study area was made with each dam assigned a rating of 1 to 5 based on their degree of passability. Locations of dams are included in the Power BI database, but were not included in the stressor analyses performed thus far. Alterations to flow (e.g., increased imperviousness that makes flows more flashy) occur locally, but can accumulate downstream as more localized pockets of imperviousness contribute to altered flows. Habitat also has a cumulative effect within watersheds, and it substantially influences aquatic life potential at the site, reach, and small watershed scales.

## IPS CONCEPTS

There have been a number of recent efforts to develop systems for assessing the restorability of streams and rivers including the U.S. EPA Recovery Potential Screening (RPS) tool for addressing CWA-related impairments (Norton et al. 2009) and the Function-Based Framework (Harmen et al. 2012) for stream restoration projects that is more narrowly focused on addressing impaired physical attributes of streams. Norton et al. (2009) provided a strong argument for the need for robust measures of restorability to avoid reliance on simple case-by-case decisions and worst-first approaches that could result in:

1. More readily restorable waters being overlooked, resulting in a lost opportunities for more optimal environmental gains;
2. Limited restoration resources can be depleted by a relatively few, severely impaired systems that may not fully recover, making it difficult to demonstrate restoration success and justify expenditures;
3. Priority-setting without a transparent and consistent basis may be vulnerable to political or legal pressures driving priorities; and,
4. The development of new and better tools and scientific knowledge about recovery may not be fully utilized in restoration decisions meant to bring about recovery.

In addition, a comparative lack of focus on protecting high quality waters is a weakness of many restoration frameworks including TMDLs. Highly susceptible waters and those that are threatened by existing and emerging stressors may well result in future degradation where the “cost-of-inaction” to protect such waters will likely result in higher costs later on or even the inability to recover them.

### The Data Driven Foundation of the IPS

The NE Illinois IPS relies on organizing a broad suite of biological, chemical, and physical parameters and indicators, the integrated analysis of which provides the underpinnings for a data-driven, scientifically sound approach to stream restoration and protection. This approach recognizes that the true economic cost of environmental management must include the remediation costs to upgrade public or private entities. These costs can be vast, for example, according to Copeland (2016): "[U.S.]EPA estimates that funding needs for stormwater management and projects to correct sewers that overflow total $106 billion over the next 20 years." Thus, the importance of using robust monitoring data and active (data-driven) adaptive management efforts to ensure that the right stressors are addressed to the correct degree is essential.

### The IPS Algorithm for Restoration and Protection

A framework for objectively sorting sites, reaches, and watersheds based on restoration potential for impairments and levels of protection for full attainment must necessarily be directed by comprehensive analyses of a robust dataset. Restorability rankings are calculated for impaired waters while Susceptibility and Threat rankings are calculated for fully attaining waters each at the site, reach, and watershed scales. While no algorithm based solely on the data will yield a framework free from the need for interpretation, a robust analysis of the data must occur prior to making informed interpretations. The algorithm used to develop Restorability and Susceptibility and Threat rankings is based on weighted ranks of aggregations of stressors, magnitudes of biological departures, and expectations for attainability with respect to the Section 101[a][2] goal of the CWA which in Illinois is the General Use. The basic assumption of this scheme is that sites, reaches, and watersheds with relatively few or no indelible stressors, less severe biological impairment, and no or reversible factors that would deter or preclude attainability are more likely to respond more completely to restoration actions than segments where the converse is true. Another key tenet is that protection of attaining waters is better than attempting restoration when they become impaired.

### Definition of Terms

The concepts of environmental Restorability, Susceptibility, and Threat are among the most fundamental outputs of the IPS framework because they provide a standardized approach to ranking existing and potential projects and taking needed actions. Definitions for each are provided in the accompanying sidebars and a brief description of the concepts for how each was quantified follows.

> **RESTORABILITY**
>
> **Restorability refers to the capacity of impaired aquatic assemblages to attain a General Use or higher condition with the application of point source controls or best management practices for nonpoint sources. Sites with high Restorability may already be close to General Use attainment and influenced by a relatively few stressors, most all of which are readily "fixable."**
>
> **Sites with lower Restorability are more likely to have more intractable stressors (e.g., concrete channels, high urban land use in *both* the watershed and riparian buffers, and multiple and severe stressor impairments).**
>
> **For site and reach-specific uses of the Restorability score it will be important to examine the suite of the most limiting stressors when developing restoration strategies.**

***Restorability***

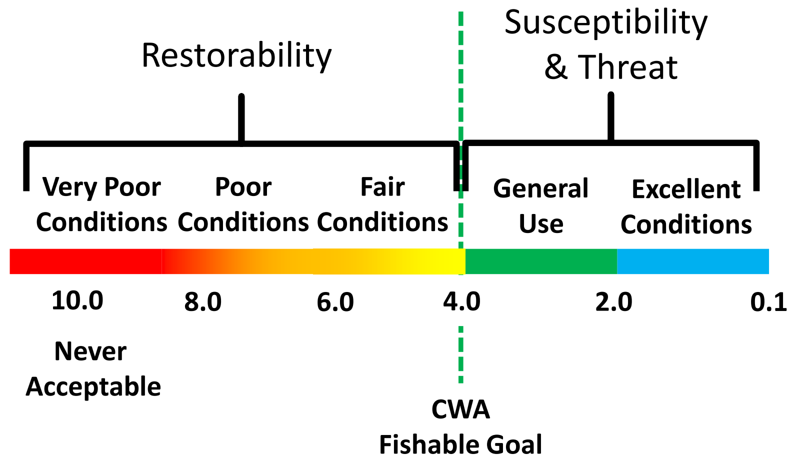



***Figure 8.*** *Schematic showing the relationship between Restorability, Susceptibility, and Threat along the scale of individual stressor and response variables and the ranges of the narrative biological condition categories from Excellent through Very Poor. General use equals good condition.*

The key goal of any water quality management program is to abate stressors enough to result in full attainment of Section 101[a][2] for aquatic life. The IPS framework targets attainment of the Illinois General Use standards and thresholds as the minimum baseline as it is consistent with the Section 101[a][2] goal. Thus when referring to Restorability, it is in deference to the General Use as the CWA baseline goal (Figure 8). For Susceptibility and Threat it is in deference to a site, reach, or watershed that is currently in Good or Excellent condition and currently attaining the General Use (Figure 8). Thus, the impetus for the NE Illinois IPS framework is restoring impairments to the General Use for aquatic life *and* protecting Excellent and Good quality waters, that latter of which is the minimum narrative condition consistent with attainment of the General Use (Figure 8).

Restorability refers to the capacity of impaired aquatic assemblages to attain the General Use or Good condition narrative. Sites, reaches, and watersheds with lower Restorability (i.e., Restorability scores >20-40) are impaired by causes that are likely more difficult to fully restore. Recovery of this degree of impairment may only be incremental and slow to respond because of the indelible characteristics of the limiting stressor(s). Sites with high and very high Restorability scores (i.e., >60-80) are more likely to be closer to attaining the General Use biocriteria and with limiting stressors that are more readily abated (e.g., conventional chemical constituents, sites amenable to habitat restoration, watersheds with more localized rather than watershed-wide impairments, etc.). For sites with intermediate Restorability scores (i.e., restorability scores >40-60) the severity and extent of the impairment within a reach or watershed and the types of limiting stressors will need to be examined in each case. Use attainability is not a direct factor in the Restorability rankings as it is in other IPS frameworks (MBI 2015) primarily because Illinois has only General Use for aquatic life that is assumed to apply in all except the very few Chicago area rivers and channels that are designated for the Secondary Contact and Indigenous Aquatic Life Use, none of which occur in the IPS study area. In order to simulate the concept of attainability factors such as the presence of channelization, the scale of modifications, impervious land cover, etc. that can limit the likelihood of restoring to the General Use have been included. Sites with very low Restorability scores may signal a need to develop a more formal approach use attainability at some point in future. More detailed associations between the Restorability score and selected biological metrics is discussed in Chapter 5.

***Susceptibility and Threat***

While many water quality management programs emphasize the tracking and restoration of impairments, it is just as important and more cost-effective to have an equal focus on protecting rivers and streams that currently meet or surpass their aquatic life goals. This would ensure that they do not become impaired over time. High quality waters provide important ecosystem services and it is these services that society values. Attaining waters already provide these services which consist of "provisioning services" such water suitable for drinking and providing food (i.e., fish, etc.), "regulating services" such as the assimilation of pollutants, carbon sequestration, and water retention in floodplains and wetlands that reduces flooding, "cultural services" such as recreation, aesthetics, and environmental education, and "supporting services" such as ecological processes associated with nutrient assimilation and sequestration (Sukhdev et al. 2010). The "Cost-of-Inaction", or in a more optimistic view, the "Benefits-of-Action", is relevant to the protection of already attaining and high quality waters, but there have been difficulties in developing measures such as *uncertainty* in estimations and *irreversibility* (OECD 2008). As applied to streams and rivers *irreversibility* may be the most important factor because it is unclear in an extensively developed urban landscape whether full restoration of aquatic life is possible beyond certain levels of watershed development particularly for the highest levels of biological condition (i.e., Excellent

**SUSCEPTIBILITY**

**Susceptibility refers to the sensitivity of <u>attaining</u> aquatic assemblages with more diverse and sensitive assemblages (e.g., higher fIBI and mIBI scores and lacking certain stressors) being the most susceptible to decline with increased stress. For the highest performing assemblages (i.e., excellent condition), the likelihood of restoring assemblages to those levels of quality may be low, thus the "*<u>cost of inaction</u>*" of not protecting such waters now will be higher later.**

**Sites attaining their goals (e.g., General Use), but with low Susceptibility scores may be more resilient because they have sensitive assemblage members, but fewer numbers of intolerant or rare taxa that are often found at sites considered to be more susceptible.**

**Sites with relatively low susceptibility scores, may however, be threatened if chemical stressors are already at levels associated with a lower level of quality (fair, poor, very poor). Threat scores are comprised of the number of elevated stressors and their severity.**

**THREATENED**

**Threatened refers to sites that are currently *attaining* their designated use biocriteria, but which have one or more stressors at levels that exceed impairment thresholds. The Threat score is low when a single stressor of low intensity, but increases as the number and/or intensity of stressors increase. Thus a site with low susceptibility, but which has pending threats should be considered a high priority for protection.**

narrative condition level). The NE Illinois streams and rivers that have the highest biological condition (i.e., Excellent fIBI and mIBI scores) and biodiversity typically have the lowest stressor "loadings" (i.e., equivalent to Excellent stressor scores $\leq$2; see Figure 8) and are considered to be the most susceptible to increased levels of stress. These sites are considered to be susceptible because stressor-response relationships predict a loss of sensitive species/taxa as stressors increase. While such sites may be considered resilient to natural stressors (e.g., flooding, drought), the knowledge that anthropogenic stressors typically result in species/taxa declines makes them susceptible. Sites that are attaining the General Use (Good), but which also have elevated stressor levels (i.e., stressor scores >2, <4), likely have some level of resiliency to the presence of stressors and have low or very low Susceptibility rankings (<20-40).

The Threat score differs from the Susceptibility score in that it focuses primarily on the presence of elevated stressors (stressor scores >4). The site or reach Threat is scored on the number of stressor categories with such elevated stressors and the magnitude of the stressor "exceedances" (i.e., fair, poor, or very poor threshold values). This means that sites that have a high Threat ranking already have elevated stressors present at more serious levels of exceedance. Thus a combined assessment of Susceptibility and Threat scores can be used to not only prioritize protection efforts, but also reveal the urgency of the need to intervene before such waters become impaired. More detailed associations between the Susceptibility and Threat scores and selected biological metrics are discussed in Chapter 5.

### Weighting of Restorability Score Factors

The factors that comprise the IPS Restorability score and its weighted factors are illustrated in Figure 9. These apply only to sites with impaired biological assemblages. It includes the fIBI and mIBI (ranked 1-10), the percentage of sites attaining the General Use biological criteria, the biological condition of sites within the same HUC12 watershed, the local habitat rank (1-10), channel condition (1-20), HUC12 watershed QHEI (1-20), catchment and riparian spatial buffer land use (each ranked 1-10), ionic strength parameters (1-15), and the number of severe (1-10) or intermediate (1-10) chemical threshold exceedances by parameter category (i.e., nutrients, metals, organics).

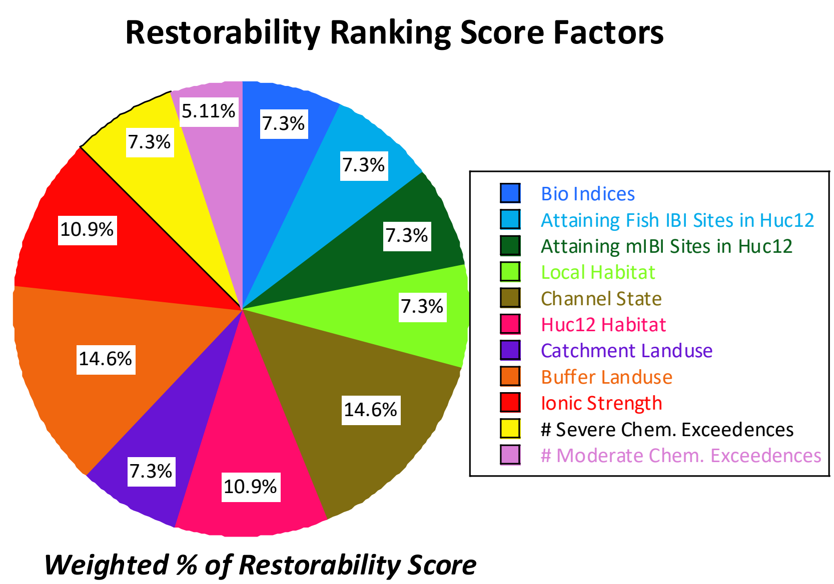



***Figure 9.*** *Maximum contribution of each of the factors that comprise Restorability rankings for impaired sites in the NE Illinois IPS study area.*

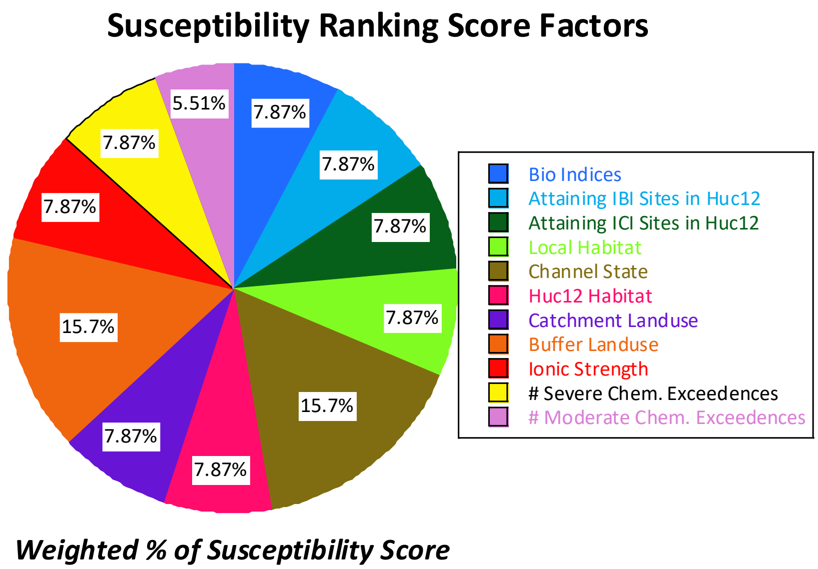



***Figure 10**. Maximum contribution of each of the factors that comprise the Susceptibility rankings for attaining sites in the NE Illinois IPS study area.*

## Weighting of Susceptibility Score Factors

Susceptibility scores are calculated only for sites that are meeting or surpassing the Illinois General Use biological criteria. A higher score indicates greater Susceptibility to an increase in stressors. Here we have relied on the fact that sensitive fish species and macroinvertebrate taxa generally decline first with increasing levels of stress and becoming more rapid as that stress increases. Thus sites and reaches with higher fIBI and mIBI scores are more susceptible to degradation because sites with higher stressor levels across multiple stressor categories have few or no sensitive species and taxa. The most biologically sensitive sites have higher Susceptibility scores because they are considered to be the most at risk to increases in stressors. Data from across the Midwest indicates that such waters have been adversely affected by the range of pollution associated with human activities and impacts. Sites that would historically rank as the highest quality and the most Susceptible (i.e., with the highest Susceptibility scores) are less common. Reference sites located outside of the core watersheds where stressors associated with increased urban development are lower and that is where we generally found the largest populations of sensitive species/taxa. Sites that are only marginally attaining the baseline General Use aquatic life use IBI thresholds and which have a low background level of stressors are considered to have a very low and low susceptibility (<20-40). The expected composition of species and taxa in streams with lower Susceptibility scores tend to be more resilient to increasing stress and they may naturally lack some of the highly intolerant species and taxa that decline or disappear under an increase in stressors. The algorithm for determining the Susceptibility score is similar to that of the Restorability score and is depicted in Figure 10. Sites that have Excellent biological assemblages have higher biological index scores, good riparian spatial buffer land uses, and the lowest stressor levels and as a result have the highest Susceptibility scores (>50). There is a similarity among several attributes within the Restorability and Susceptibility ranking algorithms with a slightly higher weighting given to natural channels and sites with more robust natural buffers in the latter.

## Weighting of Threat Score Factors

A Threat ranking that focuses on stressors that are already present is also derived for attaining sites. Threat factors and their weighting are depicted in Figure 11. Each stressor received a score of 1 if the stressor is in the fair range, a score of 3 if the stressor is in the poor range, and a score of 7 if the stressor is in the very poor range. The threat score was then normalized to a scale of 0-100 with 0 indicating no known threats and the highest threat score indicating the presence of multiple stressors ranked as poor or very poor. The Threat score can be used to identify sites that currently attain the General Use biological criteria, but which have levels of stressors that if increased any further would likely result in biological impairment. For example, a site may have a low Susceptibility score because it is a General Use designated stream that is marginally attaining the biocriteria, but which receives a higher Threat score because of elevated chemical stressors in the fair range or worse. The importance of the Susceptibility and Threat rankings is to prompt taking action before an impairment occurs, supporting a protective mode of management that should complement the traditional emphasis on restoration.

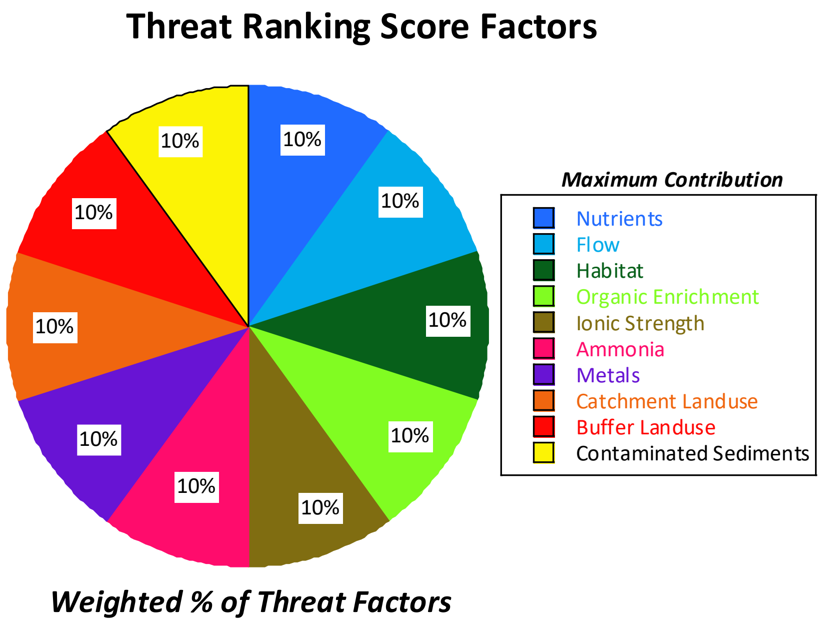



***Figure 11***. *Maximum contribution of each of the factors that comprise the Threat rankings for attaining sites in the NE Illinois IPS study area.*

In order to standardize the interpretation of complex environmental data, each with different measurement units and scales, the individual stressor and response components of the IPS are normalized to an intuitively consistent scale (Table 4). This scale is also linked to the range of narrative categories that include Excellent, Good, Fair, Poor, and Very Poor in which the Illinois General Use for aquatic life is Good and serving as the baseline restoration goal under the CWA. The Excellent range serves as a high end protection benchmark under a theoretical framework of use subcategories. The Fair, Poor, and Very Poor narratives do not meet the General Use, but the Fair and Poor ranges could also serve as theoretical use subcategories when and if formal use attainability analysis[7] is considered in the future.

Both the biological and stressor data are used to illustrate overall quality (e.g., Excellent, Good, Fair, Poor, or Very Poor quality), the severity and extent of impairments as portrayed by the degree of departure from a biocriterion, the miles of stream or river in an impaired condition, and the frequency and magnitude of stressor threshold exceedances. Based on complements of individual stressor and response results, a Restorability score is derived for all *impaired* sites

[7] Use attainability analyses need to conform to federal regulations at 40CFR Part 131.

***Table 4**. Summary of the IPS algorithm including the stressor ranking values (0.1 -10) linked to the biological narrative condition ranges, theoretical use subcategories, and coinciding ranges of Restorability, Susceptibility, and Threat scores (the latter two include intermediate (>40-60), high (>60-80), and very high (>80) scores.*

| Stressor and Response Variables (0-10 Standardized Scale) | | | Restorability, Susceptibility and Threat Score Ranges (0-100 Range) | | |
|---|---|---|---|---|---|
| **Narrative Condition** | **Theoretical Use Subcategory** | **Stressor Ranks** | **Restorability** | **Susceptibility** | **Threatened** |
| **Excellent** | **Exceptional** | **0.1-2.0** | **Not assigned to attaining sites** | **Very Low (0-20)** | |
| **Good** | **General Use** | **>2.0-4.0** | | **Low (>20-40)** | |
| **Fair** | **Modified Use** | **>4.0-6.0** | **V. High (>80)** | **Not assigned to impaired sites** | |
| | | | **High (>60-80)** | | |
| **Poor** | **Limited Use** | **>6.0-8.0** | **Intermediate (>40-60)** | | |
| | | | **Low (>20-40)** | | |
| **Very Poor** | **None** | **>8.0** | **V. Low (<20)** | | |

and a Susceptibility and Threat score are derived for sites that are *attaining* the applicable biological criteria. The Restorability and Susceptibility/Threat scores are each based on a 0-100 scale to normalize the stressor and response scales of measurement and are considered relative values. The narrative binning of the Restorability and Susceptibility and Threat scores is a placeholder pending the actual use of the scores in their pending application to watershed restoration and protection. We expect that the details about the Restorability, Susceptibility, and Threat scales will become more refined as they are applied in watershed bioassessments by the watershed groups and stakeholders for making restoration and protection decisions in the near future. The IPS Dashboard in Power BI serves as a platform for engaging and supporting users and for making improvements to the IPS framework as it is utilized in the future.

## STRESSOR DERIVATION METHODS

This section describes the analytical methods and stressor threshold derivation steps for the IPS. The thresholds are not intended to only serve as "stand alone" criteria, but to also support a lines-of-evidence stressor identification process that is accomplished within the context of recurring watershed bioassessments. Biological response signatures from the fish and macroinvertebrate assemblage data, habitat data, pollutant loadings, chemical threshold exceedances, information about sources of stressors, and field observations are integrated at the site, reach, and watershed scales by the IPS framework to determine the most limiting stressors leading to the development of protection and abatement actions. The IPS Dashboard is designed to make such data and outputs readily available to support integrated analyses by users each of whom may have different needs, goals, and objectives. The sequence of steps in the derivation of stressor thresholds and their use in a tractable stressor identification process supported by the outputs of the IPS housed in the IPS Dashboard is portrayed at the end of this chapter.

The following describes the methods, analytical techniques, and major steps in the derivation of stressor thresholds. A wide array of analytical techniques are available and have been used in numerous similar studies to develop stressor thresholds. While the techniques used herein were selected because they have performed well in other projects, those that were not used may be equally credible. By making the database available in the IPS Dashboard users can apply alternate techniques to validate the components of the IPS as developed to date. It is expected that the IPS will be refined as it is applied by the NE Illinois watershed groups and their stakeholders with additions and modifications being made as deemed necessary.

**Step 1: Stressor-Specific Species and Taxa Sensitivities: Weighted Means**

Weighted stressor values (WSVs) were used to rank and identify fish species and macroinvertebrate taxa as intolerant, sensitive, or tolerant to a particular stressor in a correlative fashion. For a "positive" attribute the value of which is highest under the best conditions such as channel condition or the QHEI score, a weighted mean is derived as a measure of central tendency where the value or score receives a high "preference" where the most individuals of a species or taxa occur at a particular value for that variable. A minimum sample size of five (5) sites was used to determine whether or not to use a WSV to classify a species or taxa as intolerant, sensitive, or tolerant for a particular parameter. The result is a field-derived Sensitive Species Distribution (SSD) for each species/taxa-group-parameter-stream size combination of data (Figure 12). The upper 20th percentile of the species/taxa rankings for a positive parameter (e.g., QHEI, D.O.) or the lower 20$^{th}$ percentile for a negative parameter (e.g., total P, total chloride) to determine which species/taxa to include as being sensitive to a particular parameter. Tolerant designations are species/taxa less than the 20$^{th}$ percentile for positive parameters or greater than the 80$^{th}$ percentile for negative parameters. Figure 12 is an SSD plot

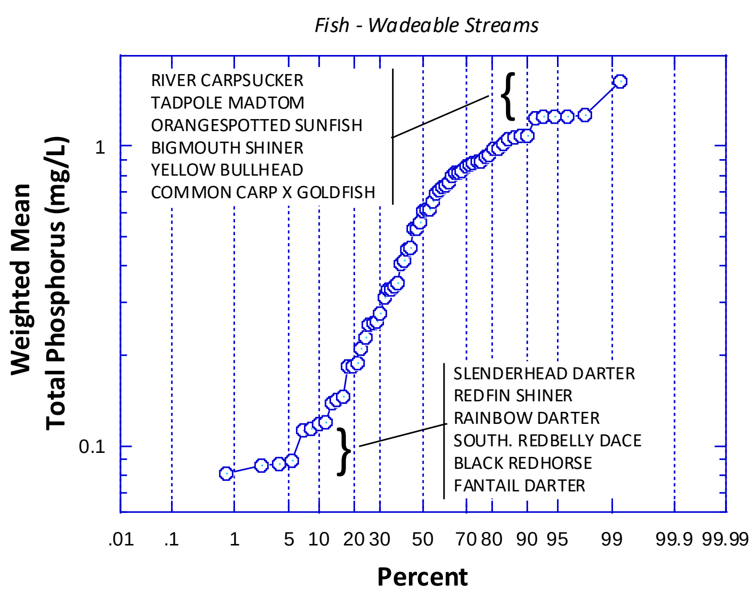



***Figure 12.*** *A field derived SSD for fish species based on weighted mean total phosphorus (TP) concentrations for IPS study area sites in NE Illinois. Selected species in the most sensitive (lower) and tolerant (upper) classifications from the distribution are labeled for illustration.*

of ranked weighted total phosphorus concentrations by fish species for sites <350 sq. mi. The stressor relationships and threshold derivation graphs and plots for the most limiting assemblage by stressor appear in Appendix B.

***Stream Size and Species Tolerances***

Many species of fish and taxa of macroinvertebrates are limited in terms of the size of stream they will inhabit. For example based on analyses by Rankin and Yoder (2010), much of which is related to habitat preferences, the number of habitat niches for fish increase with increases base flow and drainage area. In addition, the expectations for concentrations of certain parameters can also vary with stream size. Reference concentrations of nutrients, for example, increase with catchment size. To better account for natural gradients three classes of waterbodies were defined for calculating species sensitivities based on overlapping boundaries that are similar to categories used to apply the biocriteria in Ohio - headwater streams (<20 sq. mi), wadeable streams, and boatable rivers. For fish drainage area and sampling gear differences (i.e., boatable vs. wadeable) were both used and for macroinvertebrates drainage area was used as the sampling method is attenuated to stream size that generally corresponds to drainage area (Table 5). Fuzzy boundaries means that all data from each drainage category in Table 5 was used when deriving the lists of sensitive species, thus sites of 25 sq. mi. for example were used in the derivation of both the headwater and wadeable sensitive species. This was done to minimize the influence of variability in natural gradients and to account for differences in species or taxa that are expected in different stream and river size panels. When biological stressor metrics are calculated (e.g., number or percent of chloride sensitive species) it was based on species that should naturally be present in these stream and river size categories.

***Table 5****. Stream and river size panels used to derive sensitive species/taxa for the NE Illinois IPS.*

| Sampling Site Types | Drainage Area "Boundaries" | IPS Fuzzy Boundaries |
|---|---|---|
| Headwaters | < 20 sq. mi. | <20 – 40 sq. mi. |
| Wadeable | 20 - ~300[a] sq. mi. | 15-350 sq. mi. |
| Boatable | ~100[b] – 2,620[b] sq. mi. | 250 – 2,620[c] sq. mi. |

[a] Approximate upper boundary based on application of wadeable fish sampling methods in the study area.
[b] Approximate lower boundary for boatable sites in the study area.
[b] Catchment size of the Fox River as the largest river basin in the IPS study area.

**Step 2. Calculate Stressor-Specific Species and Taxa Richness for Each Sample**

Once the species and taxa were classified as sensitive or tolerant, the classifications were used to derive the species/taxa richness of sensitive species/taxa in each sample in the NE Illinois IPS database. Four plots were developed:

1) A scatterplot of the stressor-specific species/taxa richness value vs. the stressor value at each site;

2) A scatterplot of the Illinois fIBI or mIBI vs. the stressor-specific species/taxa richness value for each sample;
3) A scatterplot of the stressor-specific species/taxa richness value vs. the Illinois fIBI or mIBI; and,
4) A probability plot of stressor values by narrative category (Excellent, Good, Fair, Poor, and Very Poor) and where sensitive species/taxa exceeded the 25$^{th}$ percentile for Excellent and Good sites for each index (see Appendix B).

Sites with stressors that strongly limit species/taxa richness generally show a well-defined "wedge-shaped" threshold pattern. In contrast, sites with stressors that only weakly or do not appear to limit species/taxa richness will show a much less well defined and variable threshold response. Sites with strong differences along an IBI/sensitive species gradient will be reflected in a clearer separation in the curves in the probability plots.

Plots of the IBIs vs. stressor-sensitive species or taxa illustrate the strength of a threshold between the stressor-specific species or taxa richness and the assemblage index. A strong, steep, and continuous threshold is evidence of a strong association with the stressor of interest with measures of overall biological condition (fIBI and mIBI scores). A variable threshold or one that appears weaker (i.e., flattened) as the index value increases signals a weaker association between a stressor and the measures of biological assemblage response.

### Step 3. Derivation of Association Thresholds by Parameter in the IPS Study Area

Quantile regression was used to visually provide an estimate of the "goodness-of-fit" of the 95$^{th}$ percentile line to measure classification strength. This is a useful method for deriving effect thresholds with field data which are generally comprised of multiple stressor gradients that are difficult to isolate by more conventional means such as linear regression. Figure 13 illustrates the principle of using wedge shaped plots of data points to first determine if a meaningful relationship exists and then develop threshold responses along the gradient of effect. While statistically significant quantile regression lines could be fitted within most of the plots, the slopes were more variable compared to the same plots developed for the Southwestern Ohio IPS (MBI 2015) thus a modified approach to deriving effect thresholds and measuring the "strength" of associations was used. A 25$^{th}$ (for positive parameters) or 75$^{th}$ (for negative parameters) percentile value was used for data that attained the Illinois General Use biocriteria (Good) or Excellent fIBI or mIBI index scores and then interpolated between the minimum or maximum stressor parameter values to assign ranks from 0.1 (best) to 10 (worst). A rank of 4.0 minimally represents the General Use or Good threshold and a rank of 2.0 the Excellent threshold. Thresholds for the Fair, Poor, and Very Poor narrative condition classes are distinguished by ranks of >4.0-6.0; >6.0-8.0, and >8.0-10, respectively.

The 25$^{th}$ and 75$^{th}$ percentiles were used in lieu of a more extreme percentile (e.g., 95$^{th}$ percentile), because other studies that employed quantile regression to derive stressor effect thresholds (e.g., Bryce et al. 2010) found that values at the extremes can have wider confidence intervals. Similar percentiles (i.e., 25$^{th}$, 75$^{th}$) have been widely used to derive biocriteria and similar response thresholds for some of the same reasons. This also reduces the influence of extreme outliers and for negative stressors it represents values above which there is a greater risk or probability of serious, adverse biological effects. Stressor thresholds were derived for each parameter for river and stream sites draining less than ~350 sq. mi. and separately for fish and macroinvertebrates. Thresholds were calculated for the Excellent, Good, Fair, Poor, and Very Poor narrative ranges of the fIBI and mIBI with Good equating to compliance with the General Use biocriteria. The more stringent of the fish or macroinvertebrate thresholds are to be used as effect thresholds in watershed bioassessments and as inputs to the derivation of the Restorability, Susceptibility, and Threat scores. Additional lines-of-evidence information was provided by categorical variable reduction analyses (e.g., correlation analyses and classification and regression tree analyses) and goodness-of-fit (FIT) analyses that are designed to provide insight into the relative contribution among the broad cause categories. The results of these analyses were used to weight the factors that went into the derivation of the Restorability, Susceptibility, and Threat factors.

### *Causal Analyses and Variable Reductions Approaches*

A key aspect of deriving thresholds is the ability to distinguish variables which are strongly correlated with causal variables from ones that are only weakly correlated with causal variables, the latter being only weakly related to biological impairments. For example, there are a large number of land use variables, many of which are highly correlated with one another. A three-step process was used to identify the "best" stressor variables for identifying the mechanisms of aquatic life impairment and providing variables that will be useful in selecting and designing restoration and protection approaches and serving as readily measurable goals for stressors. For each category of stressors, a correlation matrix was developed to eliminate variables that are highly correlated (r value >0.70). Among the highly correlated variables the readily measured ones that match conceptual models and outside evidence of causal impacts were retained. Classification and regression trees were used to identify the strongest among the remaining variables in explaining IBI scores (regression) or attainment of General Use and Excellent IBI thresholds. After this initial assessment, the association between stressor rank and sensitive fish species or macroinvertebrate taxa was used to back calculate the stressor rank at each site to estimate the ranking based on observed species/taxa richness. The stressor rank based on the measured stressor was then compared with the predicted stressor rank based on observed sensitive species or taxa presence to produce a goodness-of-fit score (FIT score) to rank the stressor variables. These relationships were then used to develop a ranking of the importance of key stressors.

### *Weighting of Stressor Values*

Each category of stressor does not have an equal impact on the aquatic assemblages among the stressor categories and even among stressors within a category. Stressor thresholds (scaled from 0.1 to 10) were based on wedge-shaped relationships between each stressor and biological indicators pegged to the most stringent or appropriate of the fish or macroinvertebrate assemblage. The number of stressor-specific sensitive fish species or macroinvertebrate taxa at a site can also be used to predict a stressor rank. A comparison of the actual stressor rank can be compared to the predicted stressor rank using a FIT analysis. Stressors that are strongly limiting along such a threshold should have a relatively "tight" relationship with few outliers that exceed the predicted threshold. Weaker stressor relationships will have more outliers exceeding the threshold. The magnitude of the deviations is also important. To quantify the goodness-of-fit ("FIT"), the actual stressor rank vs. the predicted stressor rank was plotted assuming that the number of stressor sensitive species or taxa is limited by the corresponding stressor at a site. A measure that quantifies the proportion of sites by bins (integer levels) of stressor ranks and predicted stressor ranks above what is expected along the threshold line of actual (stressor level = predicted stressor level) was then derived. The magnitude of the deviation was weighted by a factor that increases by 1 the further above the predicted line (by integer categories) and weighted by the proportion of all sites represented by that bin as follows:

$$FIT = \sum_{n=1}^{Bins} (\mathrm{BPE} * \mathrm{W} * \mathrm{P})$$

*where;* BPE = %Bin Prediction Error, which is the percentage of sites in that bin compared to those samples in a measured stressor bin column;
W = Bin weighting (1-9) where weighting values increase by 1 with each bin exceeding the prediction line;
P = proportion of all sites that are represented in that bin.

Figure 13 illustrates the weighting factors on a hypothetical plot of an actual stressor rank vs. a predicted stressor rank. The lower the FIT score *the stronger is the relationship*. FIT values increase when there are a greater proportion of sites exceeding the prediction line and as the distance from the line increases resulting in higher weighted scores.

Within stressor categories the FIT value was used to rank the relative strength of each stressor relationship with biological response. For habitat stressors the rankings are; QHEI Embeddedness (0.01) >QHEI Score (0.04) >QHEI Substrate Score (0.04) >Good QHEI Attributes (0.04) >QHEI Channel Score (0.07) down to QHEI Gradient Score (0.31). Some parameters such as PAH compounds and metals had the weakest FIT scores although these can contribute to localized impairments at sites where other related stressors are also prevalent (e.g., sites with high impervious cover, heavily developed urban land use). Nutrients, are also important stressors based on FIT scores with TP (0.04) >Min. D.O. (0.10) >TKN (0.14) >$BOD_5$ (0.21) >Nitrate-N (0.29). A more detailed discussion of the FIT analyses appears in Chapter 4.

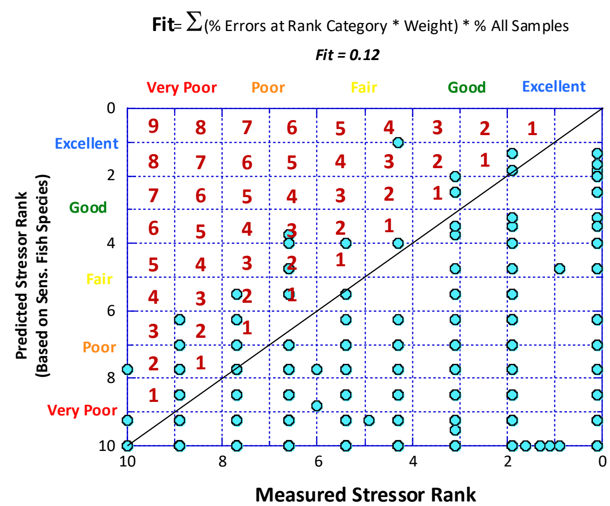



*Figure 13. Maximum contribution of each of the factors that comprise the Restorability ratings for impaired sites in the NE Illinois IPS study area.*

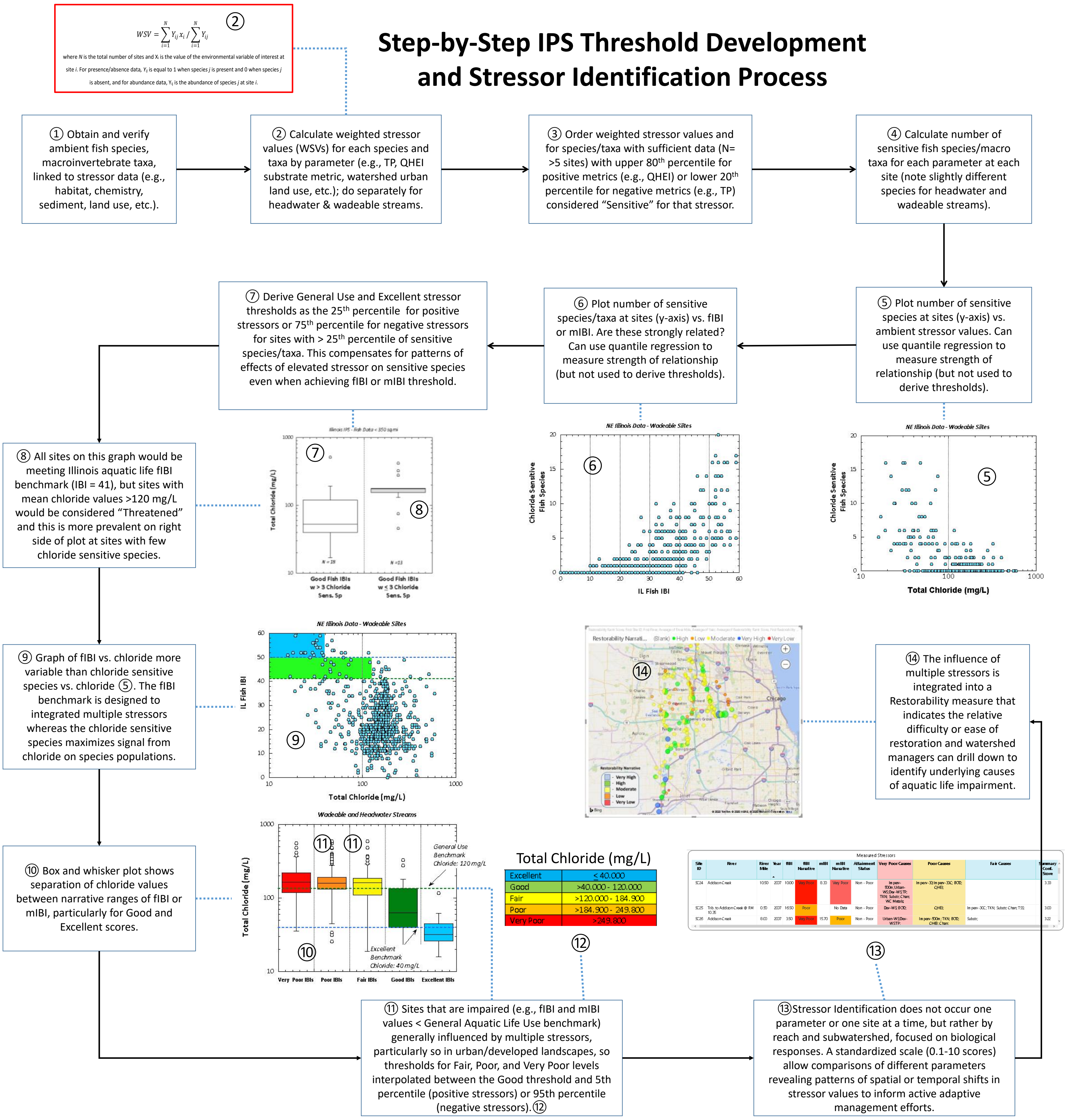



***Figure 14****. Step by step depiction of the derivation of stressor thresholds and their use in a stressor identification process supported by the outputs of the NE Illinois IPS model that are available in the IPS Power BI dashboard.*

# Chapter 3: Atlas of Stressor Relationships

## INTRODUCTION

A key aspect of an IPS is the development of regional stress:response thresholds for chemical, physical, and landscape stressors that show a logical relationship with biological responses along a Biological Condition Gradient (BCG; U.S. EPA 2015). The resulting levels of a particular stressor that correspond to a desired biological condition can then be used to evaluate the severity of causes that are associated with biological impairments. This is an improvement over the previously employed approach where causes were defined as being present and were based on thresholds that were "borrowed" from other states given the dearth of such measures that are available for Illinois rivers and streams. However, in an IPS framework such thresholds also provide the essential weighting for the development of Restorability rankings for impaired sites and Susceptibility and Threat rankings for attaining sites. Full attainment of the Illinois General Use for aquatic life is the "minimum" goal for Illinois rivers and streams.

### Regionalization of Stressor Thresholds

The identification of stressors associated with biological impairments in prior NE Illinois watershed bioassessments was sponsored by the five watershed workgroups (Appendix A). Since 2006 these assessments have relied on the water quality criteria (WQC) in the Illinois WQS, which was available only for parameters with such criteria[8]. For parameters that lack WQC and/or which are outdated, stressor thresholds from national compendia (e.g., sediment screening guidelines), statewide values (e.g., Illinois non-standard benchmarks), sufficiently scoped studies in nearby states (e.g., Ohio reference sites [Ohio EPA 1999], SW Ohio IPS [MBI 2015]), and/or those derived from the original IPS (Miltner et al. 2010) have been used. Table 6 provides a compendia of the thresholds that had been used to identify potential stressors in biological and water quality reports for the DuPage River, Salt Creek, and Des Plaines River watersheds in 2006-2019 prior to having IPS developed thresholds. For the purposes of the IPS framework the term *threshold* is used as the point at which an effect is evident in an ambient biological response, i.e., the concentration or otherwise measured level or quantity of a particular chemical, physical, or land use stressor corresponding to a change in a biological assemblage measure. This differs somewhat from the term *benchmark* which is a standard or point of reference by which a result is measured or judged. The term benchmark is more appropriate for the response of the biological assemblages indicated by fIBI and mIBI values that equate to the Illinois General Use for aquatic life and the five narrative categories of Excellent, Good, Fair, Poor, and Very Poor used herein.

Regionally derived stressor thresholds should account for the species and taxa that are resident in the rivers and streams of that region to ensure that they are not only protective, but also representative of faunal sensitivities to the stressors that are present. For the parameters that

[8] The terms criteria or criterion are used herein and consistent with the WQS being the designated use plus the criteria for protecting and measuring attainment of that use, while acknowledging that IEPA prefers the term "standard".

***Table 6.** Chemical thresholds consisting of Illinois water quality criteria, biological effects thresholds, and non-effect reference benchmarks used to support the assignment of causes to observed biological impairments in the Upper Des Plaines watershed. Only the most commonly observed chemical parameters in water column samples that have been detected in NE Illinois water samples are included.*

| Parameter[1] | Water Quality Criteria[2] | | Effect Thresholds[3] | | | | Non-effect Benchmarks[4] | |
|---|---|---|---|---|---|---|---|---|
| | Illinois Chronic | Illinois Acute | Ohio EPA[5] | SW Ohio[6] | NOAA SQRT[7] | Other | Regional Reference[8] | Illinois Non-Standard[9] |
| ***Demand Group*** | | | | | | | | |
| **$BOD_5$** | NA[10] | NA | -- | 2.48 mg/L [HW Streams] 2.96 mg/L [WD Streams] 2.60 mg/L [BT Rivers] | -- | -- | 2.00 mg/L [HW Streams] | -- |
| **Dissolved Oxygen (D.O.)** | 5.5./6.0 mg/L [7-day rolling avg.] | 3.5/5.0 mg/L [minimum] | 7.2 mg/L [HW Streams] | 5.32 mg/L [All Streams] | -- | -- | 6.6 mg/L [HW Streams] | -- |
| **Suspended Solids (TSS)** | NA | NA | 16.0 mg/L [HW Streams] | 65.7 mg/L [HW Streams] 70.8 mg/L [WD Streams] 74.3 mg/L [BT Rivers] | -- | -- | 28.0 mg/L [HW Streams] | -- |
| ***Nutrients Group*** | | | | | | | | |
| **Ammonia-N ($NH_3$-N)** | 1.24 mg/L [pH 8.0/25°C] | 8.40 mg/L [pH 8.0/25°C] | 0.05 mg/L [HW Streams] | 0.31 mg/L [HW Streams] | -- | 0.15 mg/L [DRSCW IPS[11]] | 0.025 mg/L [HW Streams] | -- |
| **Total Kjeldahl Nitrogen (TKN)** | NA | NA | 0.50 mg/L [HW Streams] | 0.51 mg/L [HW Streams] 0.58 mg/L [WD Streams] 1.05 mg/L [BT Rivers] | -- | 1.00 mg/L [DRSCW IPS[11]] | 0.70 mg/L | -- |
| **Phosphorus** | NA | NA | 0.216 mg/L [HW Streams] | 0.080 mg/L [HW Streams] 0.010 mg/L [WD Streams] 0.17 mg/L [BT Rivers] | -- | | 0.072 mg/L | 0.610 mg/L |

***Table 6.****Chemical thresholds consisting of Illinois water quality criteria, biological effects thresholds, and non-effect reference benchmarks used to support the assignment of causes to observed biological impairments in the Upper Des Plaines watershed. Only the most commonly observed chemical parameters in water column samples that have been detected in NE Illinois water samples are included.*

| Parameter[1] | Water Quality Criteria[2] | | Effect Thresholds[3] | | | | Non-effect Benchmarks[4] | |
|---|---|---|---|---|---|---|---|---|
| | Illinois Chronic | Illinois Acute | Ohio EPA[5] | SW Ohio[6] | NOAA SQRT[7] | Other | Regional Reference[8] | Illinois Non-Standard[9] |
| **Nitrate-N ($NO_3$-N)** | NA | NA | 0.90 mg/L [HW Streams] | 0.96 mg/L [HW Streams] 1.38 mg/L [WD Streams] 1.68 mg/L [BT Rivers] | -- | | 1.87 mg/L [HW Streams] 1.80 mg/L [EPA Ecoregion 54] | 7.80 mg/L |
| ***Ionic Strength Group*** | | | | | | | | |
| **Chlorides** | NA | 500 mg/L; | 46.0 mg/L [HW Streams] | 52.6 mg/L [HW Streams] 59.1 mg/L [WD Streams] 68.4 mg/L [BT Rivers] | -- | 112 (fish); 141 (macro.) mg/L [DRSCW IPS[11]] | 35.0 mg/L [HW Streams] 31 mg/L (WD Streams) 55 mg/L [BT Rivers] | -- |
| **Conductance, Specific** | NA | NA | 966 µS/cm [HW Streams] 861 µS/cm [WD Streams] 770 µS/cm [BT Rivers] | 703 µS/cm [HW Streams] 660 µS/cm [WD Streams] 730 µS/cm [BT Rivers] | -- | 300 µS/cm [EPA draft[12]] | 751 µS/cm [HW Streams] | -- |
| **Dissolved Solids (TDS)** | NA | 1500 mg/L [Dec. 1-Apr. 30; expires 2018] | -- | 364 mg/L [HW Streams] 384 mg/L [WD Streams] 395 mg/L [BT Rivers] | -- | -- | 296 mg/L [SW Ohio HW] | -- |
| **Sulfate** | 1809 mg/L | -- | 334 mg/L [HW Streams] | 119 mg/L [HW Streams] | -- | -- | 118.8 mg/L [HW Streams] 120 mg/L [WD Streams] 115 mg/L [BT Rivers] | -- |

***Table 6.*** *Chemical thresholds consisting of Illinois water quality criteria, biological effects thresholds, and non-effect reference benchmarks used to support the assignment of causes to observed biological impairments in the Upper Des Plaines watershed. Only the most commonly observed chemical parameters in water column samples that have been detected in NE Illinois water samples are included.*

| Parameter[1] | Water Quality Criteria[2] | | Effect Thresholds[3] | | | | Non-effect Benchmarks[4] | |
|---|---|---|---|---|---|---|---|---|
| | Illinois Chronic | Illinois Acute | Ohio EPA[5] | SW Ohio[6] | NOAA SQRT[7] | Other | Regional Reference[8] | Illinois Non-Standard[9] |
| ***Metals Group*[13]** | | | | | | | | |
| **Arsenic (As)** | 0.190 mg/L | 0.360 mg/L | 0.002 mg/L [HW Streams] | -- | 0.190 mg/L [Chronic] | See SQRT | 0.001 mg/L [HW Streams] | -- |
| **Copper (Cu)** | 0.022 mg/L | 0.036 mg/L | 0.010 mg/L [HW Streams] | 5.9 µg/L [HW Streams] 8.9 µg/L [WD Streams] 10.4 µg/L [BT Rivers] | 0.009 mg/L[C] 0.130 mg/L[A] | See SQRT | 5.0 µg/L [HW Streams] 5.0 µg/L [WD Streams] 5.0 µg/L [BT Rivers] | -- |
| **Lead (Pb)** | 0.051 mg/L | 0.245 mg/L | 0.002 mg/L [HW Streams] | 2.7 µg/L [HW Streams] 17.4 µg/L [WD Streams] 26.8 µg/L [BT Rivers] | 0.0025 mg/L[C] 0.065 mg/L[A] | See SQRT | 2.5 µg/L [HW Streams] 2.5 µg/L [WD Streams] 3.0 µg/L [BT Rivers] | -- |
| **Manganese (Mn)** | 3.52 mg/L | 8.15 mg/L | 0.942 mg/L [HW Streams] | 98 µg/L [HW Streams] 347 µg/L [WD Streams] 472 µg/L [BT Rivers] | 0.080 mg/L[C] 2.300 mg/L[A] | See SQRT | 0.185 mg/L [HW Streams] | -- |
| **Zinc (Zn)** | 0.073 mg/L | 0.273 mg/L | 0.010 mg/L [HW Streams] | 16.4 µg/L [HW Streams] 39.3 µg/L [WD Streams] 60.8 µg/L [BT Rivers] | 0.120 mg/L [Chronic] | See SQRT | 15 µg/L [HW Streams] 15 µg/L [WD Streams] 20 µg/L [BT Rivers] | -- |

**Table 1 Footnotes**: Parameter values as total unless specific otherwise. [2] Illinois WQS (Illinois Administrative Code Part 302) - http://www.epa.illinois.gov/topics/water-quality/standards/derived-criteria/index. [3] Field-based thresholds using fish & macroinvertebrate endpoints; [4] Represent analyses of large scale ambient chemical databases with statistical approaches. [5] Biocriteria derived threshold values (2 Interquartile Ranges [2IQR] above median) in *Appendices to Association Between Nutrients and the Aquatic Biota of Ohio River and Streams* (Ohio EPA 1999). [6] Biological assemblage effect thresholds derived for SW Ohio in *Integrated Prioritization System (IPS) Documentation and Atlas of Biological Stressor Relationships for Southwest Ohio* (MBI 2015). [7] NOAA Screening Quick Reference Tables (SQRT; NOAA 2008) – hardness dependent parameters at 100 mg/L hardness; with EPA EcoUpdate Ecotox Thresholds EPA/F-95-038. [8] Ohio regional reference values (2 Interquartile Ranges [2IQR] above median) in Ohio EPA (1999). [9] 1 and 2 standard deviations (SD) above the mean of all values measured statewide. [10] NA –not included in Illinois WQS.[11] DRSCW IPS derived threshold. [12] U.S. EPA field-based threshold for Central Appalachian streams in U.S. EPA (2011). [13] Hardness dependent metals shown at 300 mg/L total hardness – see IAC Part 302 for formulae.

do not have a state aquatic life criterion (e.g., nutrients, habitat, bedded sediments, certain ionic strength parameters, outdated criteria), the application of national or even statewide criteria could be over or under protective. Today, some of the most problematic stressors are comprised of naturally occurring constituents that have optimum levels for aquatic life, but can lead to an impaired condition if they become too elevated (e.g., chloride) or depressed (e.g., habitat). For such parameters, regionally derived thresholds can better account for differences in stream types (e.g., watershed size, gradient) and be more representative than ones derived at too large of a spatial scale (e.g., national, statewide). As a result, the NE Illinois IPS stressor thresholds have effectively replaced most of those listed in Table 6 and have been used in the periodic rotation of watershed level biological and water quality assessments since 2020.

## STRESSOR AND RESPONSE VARIABLES

To achieve consistency across multiple stressor and response variables that vary in their respective units of measurement, each was normalized to a 0-10 scale along a gradient of condition from Excellent to Very Poor. All variables were ranked from 0.1 to 10 with 0.1 being equivalent to the highest quality condition and 10 the lowest quality condition (see Table 4). This approach also standardizes each variable along the narrative condition categories reflected in the General Use (good narrative) and excellent narrative fIBI and mIBI benchmarks. Blue shaded results represent conditions consistent with the Excellent narrative and green shaded results are consistent with the General Use or Good narrative benchmark. The Fair (yellow), Poor (orange), and Very Poor (red) narrative benchmarks represent increasing departures from the General Use for aquatic life which in Illinois represents the minimum goal of the CWA (Section 101[a][2]).

Exceedances of individual stressor thresholds do not always coincide with or portend a biological impairment. Sites that meet the biological criteria for the General Use for aquatic life, but which have exceedances of stressor thresholds are likely to have intermediate, high, or very high Threat scores with few or no Poor or Very Poor exceedances. A Threatened status means that the probability of biological impairment could increase with further increases in the magnitude and severity of stressor threshold exceedances which also becomes more likely when more than one stressor deviates from an impairment threshold. By ranking stressors in accordance with their likely influence on aquatic life, i.e., as Fair, Poor, or Very Poor exceedances, it makes comparisons of values from site to site, reach to reach, and watershed to watershed more meaningful.

The spatial density of the sampling locations employed in the intensive pollution survey design also allows for the consideration of the spatial extent and severity of reach-scale impacts that might limit or interfere with biological recovery or attainment. It also better quantifies the status of the biological responses which results in a more accurate and robust foundation for assigning priorities for protection and restoration. It also reduces the degree of extrapolation of the results to unsampled sites and reaches by filling in gaps left by coarse scale monitoring designs. The goal of the IPS framework is to allow users to better identify the extent and severity of impairments, identify the most limiting factors, and determine whether they coincide with the protection and restoration priorities and plans of each of the watershed groups and what other interested stakeholders might deem as being important.

## Stressor Threshold Derivation

The derivation of the Restorability, Susceptibility and Threat scores that are a major output of the IPS framework are dependent on the derivation of thresholds for the most limiting stressors to aquatic life in the watersheds of the NE Illinois IPS region. Identifying incorrect limiting stressor(s), or simply using "off-the-shelf" indicators (e.g., TSS as *the* indicator for MS4 programs) provides a weak foundation for decision making in support of protection and restoration options undermining the likelihood that they will have the intended effect. An important component of the IPS is the "just beneath the surface" analyses that help to identify the key limiting stressors to aquatic life and hence the attainment of the biocriteria benchmarks applicable to NE Illinois streams and rivers. The remainder of this Chapter documents the derivation of stressor thresholds and the identification of the most limiting stressors for streams and rivers in the NE Illinois IPS study area. This process builds on and improves the previously used stressor thresholds (Table 6) and the stressor identification process that has been conducted in the individual watershed assessments through 2019.

***Table 7**. Major categories of stressors and example parameters used in the derivation of stressor thresholds for the NE Illinois IPS.*

| Stressor Category | Example Parameters |
|---|---|
| **Physical Habitat** | QHEI and metrics, Hydro-QHEI, watershed scale habitat |
| **Nutrients** | TP, nitrate, TKN, Max. DO, diel DO Flux. |
| **Organic Enrichment** | DO, $BOD_5$, total ammonia, TKN |
| **Dissolved Materials** | Chloride, sulfate, sodium, conductivity, TDS |
| **Suspended Materials** | TSS, VSS, Turbidity |
| **Water Column Toxicants** | Metals, organics |
| **Sediment Toxicants** | PAHs, metals, PCBs |
| **Catchment Land Use** | Impervious surface, Developed land uses, road density |
| **Spatial Buffer Land Use** | Impervious surface, Developed land uses, road density |

### *Non-biological Stressors*

Although there are literally hundreds of individual stressors that exist in the urban aquatic environment (Bradley et al. 2023), they can be partitioned into categories of stressors. The most prevalent categories in the streams of NE Illinois that are likely to contribute to aquatic life impairment and threat are listed in Table 7. For example, sediment toxicant thresholds for heavy metals and major polycyclic aromatic hydrocarbon (PAH) compounds were able to be derived because they were measured at varying levels more frequently than many other sediment parameters, most of which were only sporadically detected or not at all. The latter precludes the derivation of reliable regional thresholds. When such sporadically occurring parameters are detected, they are best dealt with on a case-by-case basis. If they become elevated on a more widespread basis based on the feedback provided by iterative monitoring and assessment, then regional thresholds could be developed. The derivation of IPS stressor thresholds for the more commonly detected heavy metals and PAH compounds also allowed comparison to the previously used consensus guidelines (e.g., MacDonald et al.

2000; Persaud et al. 1993) to local waters. The same applies to the water column parameters that are commonly detected in NE Illinois rivers and streams and those that are rarely detected.

### *Measuring Biological Response*

The fIBI and mIBI are multimetric indices that Illinois uses to measure attainment and non-attainment of the General Use for aquatic life (IEPA 2018, 2020, 2022) hence they are the established arbiters of aquatic life use status for Illinois. These types of indices are designed to integrate the effects of all stressors, partly by having individual metrics comprised of species and taxa attributes that respond in a predictable manner along different parts of the stressor gradient and specifically to different categories of stress (habitat, toxics, nutrients, dissolved solids, etc.). Two assemblage groups are used in Illinois (fish and macroinvertebrates). These groups may respond differentially to the same stressors (e.g., Marzin et al. 2012) such that one index may be attaining its biocriteria while the other reveals an impairment. This is consistent with the U.S. EPA (2013) bioassessment program evaluation methodology.

The IEPA bioassessment program underwent a series of such evaluations between 2002 and 2012 using the Critical Elements Evaluation (CEE) process (Yoder and Barbour 2009). Soon thereafter the CEE was documented in a U.S. EPA methodological document entitled *Biological Assessment Program Review: Assessing Level of Technical Rigor to Support Water Quality Management* (U.S. EPA 2013). While a number of opportunities for improving the level of rigor of the IEPA program were identified (MBI 2010, 2013), the fIBI and mIBI were found to be capable of assessing Illinois rivers and streams beyond a pass/fail basis. In terms of their respective critical technical elements scoring, both Illinois and Ohio scored 3.5 and 4.0, respectively, for the ecological attributes and discriminatory capacity elements which is at or near the maximum score of 4.0 (MBI 2010). The approach of using a fully calibrated and regionally relevant IBI fulfills one of the originally intended purposes of Karr et al. (1986) to assess *". . . large numbers of sample areas and to determine trends, thus enabling us to assess the effects of management programs for water resources . . ."* . It also reflects the unique role of an IBI for which no suitable surrogate exists. The principal deficiency in the IEPA/IDNR fish assemblage methodology is the omission of DELT anomalies. However, this attribute is used as a metric used in the vast majority of fish IBIs has been routinely collected and analyzed in the NE Illinois fish assemblage assessments supported by the five watershed groups since 2006.

The statistical properties of the Illinois fIBI was examined by Gerritsen et al. (2011) who found the coefficient of variation at least disturbed sites was 9.5%, but was higher at impaired sites, which is not to be unexpected. Holtrop and Dolan (2003) analyzed the precision of the fIBI as the mean difference in resampled sites which was 17% or 10 fIBI units on a 60-point scale. The Illinois IBI has similar structural properties to the Ohio IBI (Ohio EPA 1987) which Fore et al. (1993) concluded reliably scales to six condition categories and with sufficient numbers (>200) of fish in a sample produces a variance of only ±2 IBI units. Thus, using the five narrative condition categories defined by Smogor (2005) for the fIBI to provide a framework for deriving tiered stressor thresholds is appropriate.

***Sensitive Species Distributions (SSD)***

Because the fIBI and mIBI are designed to integrate the effects of all stressors that are present, the aggregate index value alone has limited value in stressor identification (Vadas et al. 2022). Identical IBI scores can be the product of entirely different stressors, which some have erroneously cited as an inherent liability. In acknowledgment of the limitation of an IBI score alone to reveal specific stressors, fish species and macroinvertebrate taxa based responses to individual stressors were developed first, then linked back to the fIBI or mIBI narrative tier and then used to develop a compendia of stressor thresholds for use in watershed bioassessments. The technical process for deriving these thresholds is termed Sensitive Species Distributions.

There are a number of ways by which effect thresholds have been derived for various stressors and each has its advantages and limitations. For many of the most common toxic pollutants, laboratory derived toxicity testing has been the conventionally accepted approach for deriving water quality criteria. The goal of this approach is to derive the concentration of a pollutant that is protective of representative species/taxa, that is assumed to protect 95% of all species, including untested ones, for a general class of waters (i.e., freshwater or marine; Stephan et al. 1985). In developing a criterion, a curve is fit to ranked toxicity data and a value is generated that represents a parameter value that will protect the most sensitive of the tested species. An advantage of this approach is that it is based on experimental data derived under controlled conditions (e.g., untreated control tests, standard temperature, water hardness, pH, etc.). A disadvantage is the uncertainty about whether the results are ecologically and/or environmentally relevant. For example, other substances present in the ambient environment could interact with a stressor in an additive, synergistic, or antagonistic manner resulting in under or overly protective thresholds. However, traditional water quality criteria are assumed to protect ~95% of all species in a region or class of waters, but they cannot account for different complements of species and taxa that reflect different levels of assemblage sensitivity. Naturally occurring factors, some of which can be unrelated to chemical activity, could reduce or amplify the effects of a pollutant leading to under or over-protective criteria. This is a particularly vexing issue with naturally occurring parameters and substances (e.g., nutrients, ionic strength compounds, sediment, attributes of habitat) where natural background factors (e.g., soils, stream size, ecotype, gradient, base flows, etc.,) can influence the exposure regimen (magnitude, exposure, and fate) of such parameters. The application of water quality criteria for toxicants, however, has contributed much to the documented improvement in ambient aquatic assemblage conditions via pollution controls. This is especially true for the discharge of pollutant loads from point sources on a water quality basis (Yoder et al. 2005, 2019; Happel and Gallagher 2021) that were resolved via point source regulation. The apparent success of applying water quality criteria for common pollutants such as biochemical oxygen demand (BOD), ammonia-N, and common heavy metals has in itself validated how those water quality criteria have been applied, the majority via NPDES permitting.

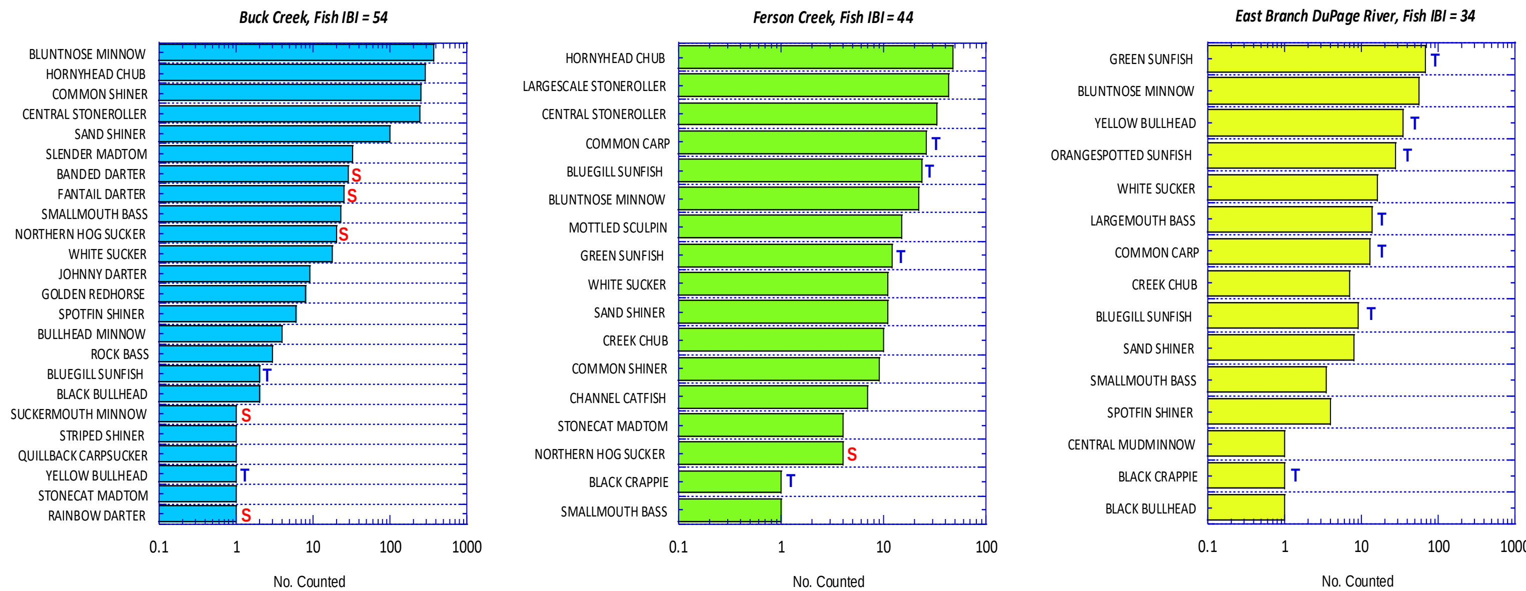
While the CWA has led to substantial progress with reducing many of the most toxic and gross water quality impairments dating between the 1960s and 1980s, many impaired waters still remain due to causes that are just now being understood. Today, the uncertainty lies with controlling pollutants *and* non-pollutants via mechanisms such as TMDLs, watershed management plans, and stormwater permitting. These impairments remain due to inadequacies (and even inaccuracies) in delineating impairments (including causes of impairments) and the unreliability of, what are in many cases, outdated water quality criteria for both historical and emerging stressors. This is one of the major reasons that the watershed workgroups in NE Illinois chose to sponsor comprehensive watershed bioassessments (U.S. EPA 2007). The stressors that are limiting to aquatic life present-day oftentimes do not have water quality criteria (*e.g.,* nutrients) or are non-toxic in their mode of effect (e.g., nutrients, bedded sediments, siltation, habitat, altered flow regime). Another strength of field derived SSDs is that they align with the concept of use attainability such that thresholds can be derived for multiple narrative condition categories that are precursors for theoretical aquatic life use subcategories. States with codified use subcategories such as Maine, Minnesota, and Ohio can discriminate between streams and rivers with the highest biological condition (Class AA in Maine, Exceptional Warmwater Habitat [EWH] in Minnesota and Ohio), those that minimally meet the CWA Section 101[a][2] goal (Class B & C in Maine, Class 2A in Minnesota, Warmwater Habitat in Ohio), and those with documented attainability limitations (Modified Warmwater Habitat in Minnesota and Ohio, Limited Resource Waters in Ohio). Traditional applications of water quality criteria do not distinguish between these classes of waters resulting in a one-size-fits-all approach. The SSD approach used to develop stressor thresholds for the NE Illinois IPS utilized the narrative condition categories of the fIBI and mIBI to simulate a theoretical use subcategory approach for both pollutants *and* non-pollutants.

**Species and Taxa Based Thresholds**

In the Region V states[9] (including Illinois) least impacted reference conditions are used to represent attainable conditions that reflect a range of quality between the Section 101[a][2] interim CWA goal for the protection and propagation of fish, shellfish, and wildlife to higher quality conditions that theoretically includes full biological integrity (MBI 2010). For the two Region V states (OH, MN) with refined ALUs in their WQS, use attainability analyses (UAAs) are routinely used to decide which use tier applies to a given reach of a river or stream. This fits well with a key premise of IBI based bioassessment approaches that species or taxa assemblage differences occur along gradients of biological condition that reflect narrative ranges of condition e.g., Excellent→Good→Fair→ Poor→Very Poor which can form the “boundaries” or benchmarks for theoretical use subcategories. While Illinois does not have use subcategories formalized in the Illinois WQS, the narrative ranges of the fIBI and MIBI can be used to simulate use subcategories for the purposes of the NE Illinois IPS.

The essential starting point for developing IPS stressor thresholds are the species and taxa sensitivities as illustrated in Figure 15. Each of the plots shown in this graph were randomly selected, one from each narrative range of the fIBI from Excellent to Very Poor. Each bar represents a species and species richness varies between each of the narrative categories. Species are coded based on their IPS-derived sensitivity to total phosphorus (TP) as measured

[9] Illinois, Indiana, Michigan, Minnesota, Ohio, and Wisconsin.



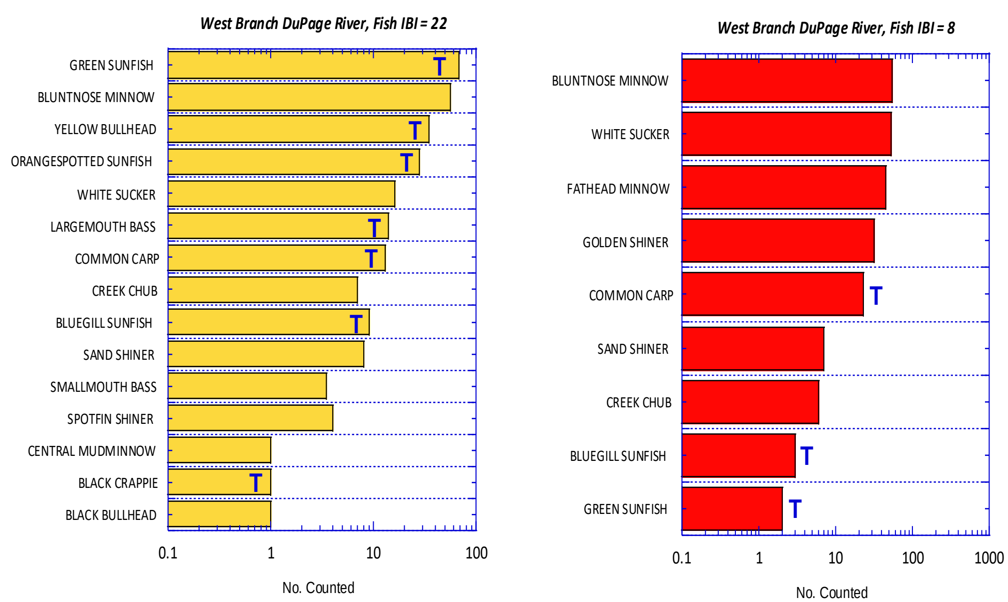




***Figure 15**. Histograms of fish species ranked by relative numbers counted at five sites in NE Illinois that represent the range of fish IBI narratives (excellent, good, fair, poor and very poor) for total phosphorus (TP). Drainage areas of the sites are between 40-60 sq. mi. TP sensitive (S) and TP tolerant (T) species are labeled.*

by weighted mean TP values (tolerant – red; sensitive – blue). In this graph, sites with Excellent fIBIs, which represents the highest quality aquatic life narrative condition range, have more species than the lower narrative ranges and there are also more species that are sensitive to TP in the Excellent range. Conversely, the sites with lower fIBI scores have fewer or no TP sensitive species and more TP tolerant species.

**Precedents for Using Field Data to Derive Stressor Thresholds**

While the majority of water quality criteria (especially for toxicants) have traditionally been derived from controlled laboratory toxicity studies (Stephan et al. 1985), effect thresholds have increasingly been derived using field data and with a variety of methods (Posthuma et al. 2003 Cormier and Suter 2013). Cormier and Suter (2013) point out that many stressors *". . . are not amenable to toxicity testing (e.g., habitat, nutrients, suspended and bedded sediment, dissolved ions), do not operate via a 'toxic' mode of effect, and that laboratory tests cannot replicate the full range of ambient exposures, effects, and interactions . . ."* observed in the field. Correlative thresholds such as those derived herein are not necessarily suitable to function as traditional water quality criteria as Cormier and Suter (2013) state *". . . that a benchmark (threshold) differs from criteria or standards in that it is not mandated by regulation, but it does provide scientific information to support decision making in various contexts."* Such thresholds are meant to be used as complements to biological benchmarks in a risk-based approach to prioritizing abatement actions for restoration and protection.

Such thresholds are not "bright-line" criteria, but rather are intended to be informative in a risk-based approach to identifying probable causes of impairment and threats to future attainment in an "active" adaptive management process. The capability to describe the magnitude of a stressor (Excellent, Good, Fair, Poor, or Very Poor) provides some advantages when conducting a lines-of-evidence stressor assessment as opposed to a bivariate pass/fail approach. For example, it is possible that a stressor such as chloride could exceed its effect threshold at a biologically attaining site and perhaps be rated as "fair" in terms of magnitude of effect. For an attaining site, the chloride exceedance would contribute to the Threat score. That score would increase as more parameters exceed their respective effect thresholds or the magnitude of exceedance is greater (e.g., very poor or poor vs. fair). Conversely, and more commonly, a stressor may not be exceeded at an impaired site (e.g., stressor level rated good or excellent) which will factor into the Restorability score along with the condition of other stressors and stressor categories. Factors that are more weakly associated with the biota on the basis of the FIT test or regression tree analysis (see Chapter 4) will receive less weight in the derivation of a Restorability score and will require a more severe threshold exceedance to be deemed a principal causative factor in an observed biological impairment.

The intent is to use such thresholds in combination with other analyses (e.g., site-specific stressor analyses and broader scale statistical analyses such as classification and regression trees (see Chapter 4) in a lines-of-evidence approach to identify the most limiting causative stressors. This is exemplified by the "combined criterion approach" that some states are taking with nutrients where both causal and response variables are used to determine when the *effects of nutrient enrichment* (e.g., increased diel D.O. swings) have become unacceptably limiting to aquatic life. Another important difference with the NE Illinois IPS is the inclusion of a much wider range of stressors than is typically included in water quality surveys and TMDLs. The IPS approach includes important co-occurring and, in some cases, inter-dependent stressors such as habitat, toxicants, dissolved materials, sediment contamination, land use data, etc. This more comprehensive approach is enabled by the watershed focused intensive pollution survey sampling design that allows data to be scaled so that cumulative effects, more detailed longitudinal patterns, and a stronger lines-of-evidence approach is available prior to developing and "locking in" regulatory actions. When executed in a systematic rotating basin approach it allows for the more efficient adaptation of BMPs and other abatement actions in response to observed changes in conditions.

Field derived thresholds can also be scaled to levels of "protection" within a hierarchy of biological potential and condition. The Biological Condition Gradient (BCG; Davies and Jackson 2006; U.S. EPA 2015) represents a framework about how biological condition (y-axis) based on a model of 10 structural and functional attributes responds to multiple stressor gradients (x-axis) that exist in a particular aquatic ecotype. Figure 16 illustrates the BCG model with the response of biological condition (y-axis) to the effect of various stressors and their thresholds along the x-axis. In this graphic, multiple stressors that occur along the x-axis can be related to their general relationship to the five narrative categories of biological condition used in the IPS that is measured along the y-axis. The "as naturally occurs" condition likely exists in only a scant few places in the U.S., but it does represent the virtual absence of stressor effects associated with pristine conditions and serving as the anchor of BCG. The intersection of the Section 101[a][2] "Fishable Swimmable" threshold is consistent with the level of protection offered by the General Use for aquatic life in Illinois and also representing the Section 101[a][2] goal of the CWA. Biological condition above these CWA boundaries generally represents minimally disturbed conditions to which a higher level of protection could be assigned, which is how the Excellent narrative range of the fIBI and mIBI is applied in the NE Illinois IPS. Minimally to least impacted reference benchmarks include the range from Excellent to Good. Traditional water quality criteria have typically been assumed, at a minimum, to protect for the Section 101[a][2] goal which is the good narrative range in the IPS. However, this is uncertain for parameters with outdated criteria. The Excellent stressor thresholds derived for the IPS are intended to apply to higher quality sites that reflect that narrative assigned to the fIBI and mIBI. The Illinois non-standard benchmarks exist for only a few parameters, and being based on data from the entire realm of Excellent to Very Poor conditions, are not reference or effect based and likely represent the lower condition categories.

It is recognized that biological expectations will vary between regions and by the level of disturbance in a particular region (i.e., background stressor condition) and as anchored in the reference condition. Stoddard et al. (2006) defined categories of reference sites that include undisturbed, minimally disturbed, least disturbed, and best attainable. The latter is where the attainability of the Section 101[a][2] goal of the CWA is hampered by legacy stressors. An example of best attainable are the biological criteria for the historically and broadly altered

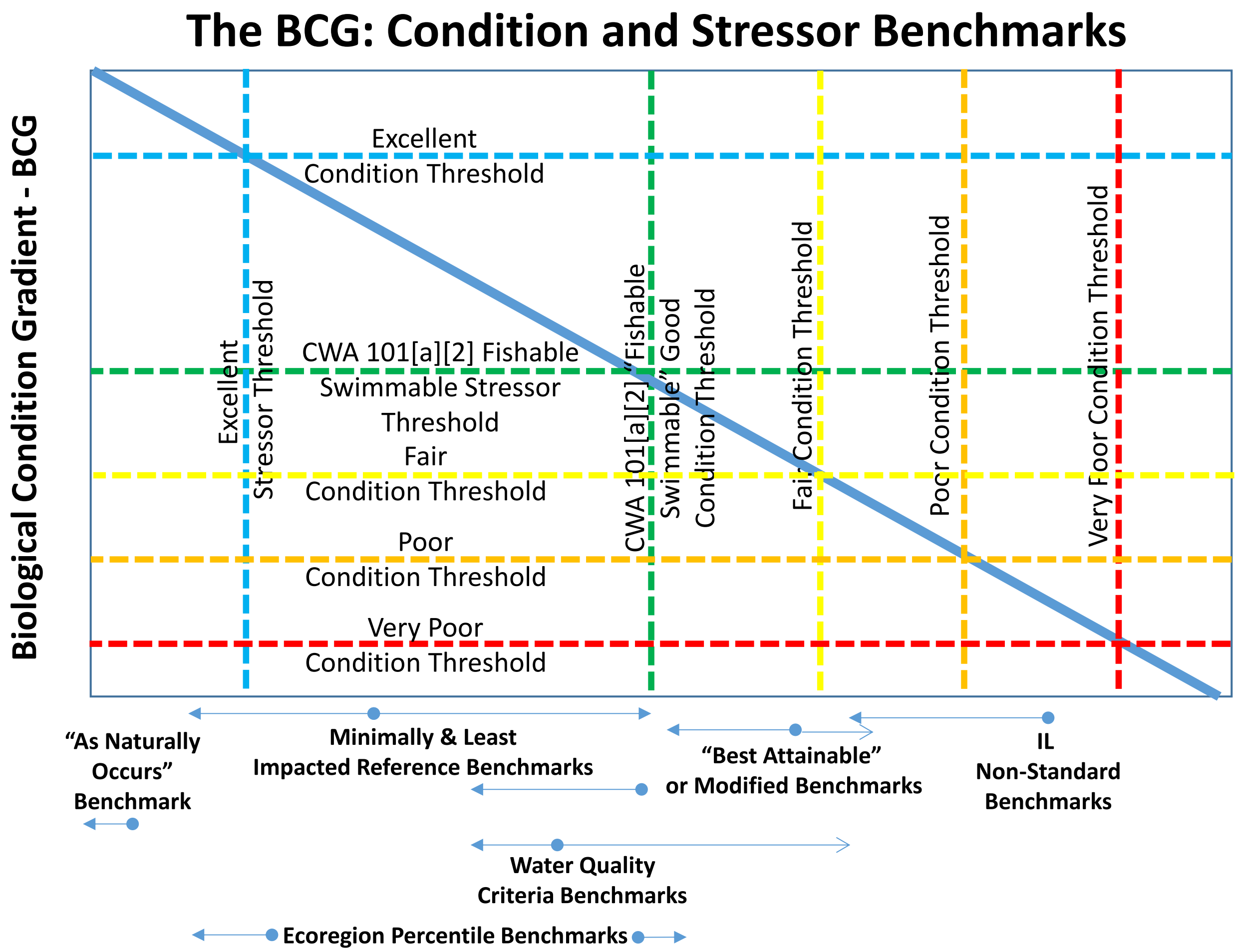



***Figure 16****. Conceptual model linking the Biological Condition Gradient (BCG: U.S. EPA 2016) to various stressor benchmarks and criteria used in NE Illinois stream and river assessments. The colored dashed lines correspond to the position of the five narrative ranges used to depict biological condition and levels of stressors. Green (Fishable Swimmable Stressor Threshold) is equivalent to GOOD and the General Use Criteria for Aquatic Life.*

Huron/Erie Lake Plain (HELP) ecoregion of Northwest Ohio (Ohio EPA 1987). This consideration resulted in a lower biological expectation (i.e., best attainable) where even that benchmark is only infrequently attained. The Modified Warmwater Habitat (MWH) aquatic life use has been widely assigned via a UAA process. While this level of widespread landscape disturbance is not as apparent in the NE Illinois IPS area, similar legacy modifications and alterations do exist and defining where stressor thresholds, to be applied to individual streams and rivers, occur along the BCG remains a guiding principle of the IPS framework.

***Limitations of Field Derived Thresholds***

Although there are many advantages to using field data to derive water quality thresholds there are also some inherent limitations. For nutrients, one limitation is the representativeness of periodic grab samples as indicators of the nutrient regime to which organisms are exposed. The data used in the IPS is generally summer-fall "normal" low flow grab sample data that attempts to measure concentrations during the most stressful period for aquatic life. The two primary nutrient parameters, nitrate ($N0_3$-N) and total phosphorus (TP), each respond differently in relation to elevated runoff. Figure 17 illustrates the responses of concentrations of TP (upper) and $N0_3$-N (lower) to a storm event in an agricultural landscape in Ohio (Ohio EPA 1999, modified from Baker 1985). TP peaked quickly and then declined rapidly while $N0_3$-N, which is delivered mostly through drainage tiles, peaks later than TP, in this example more than 12 days after the runoff event (Figure 17, bottom). Thus for $N0_3$-N, a summer normal low flow grab sample may reflect elevated levels for several days after an event occurred, and it may not reflect more frequently occurring concentrations under sustained low flows. It does measure the "what gets left behind" aspect of nonpoint source runoff.

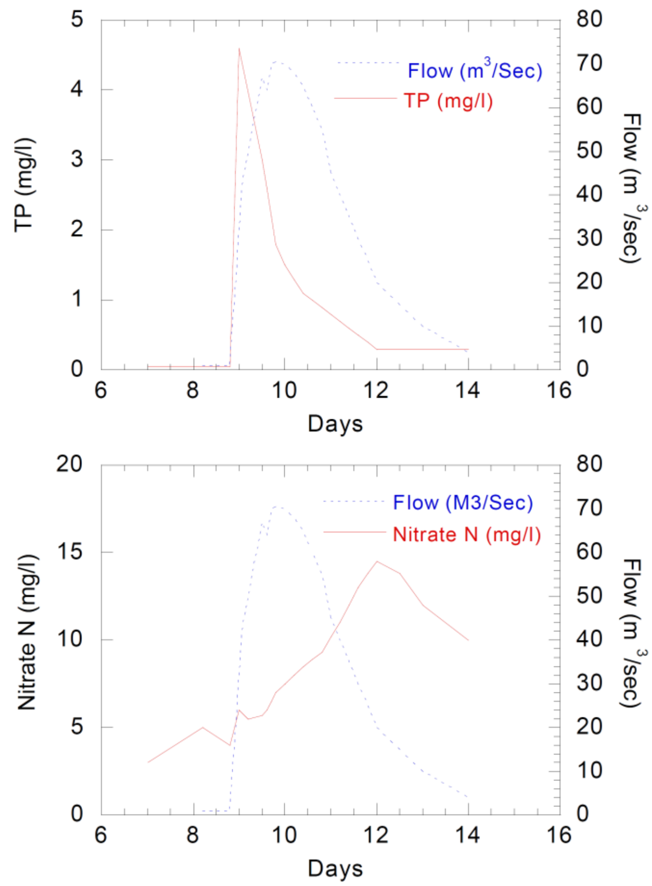



***Figure 17.*** *Plots of TP and nitrate-N vs. days since a storm event (after Baker 1985 and Ohio EPA 1999).*

Other parameters such as D.O can fluctuate diurnally. Here, lower early morning values reflect algal respiration at night and elevated late afternoon values reflect algal photosynthesis during the day. Ambient pH values fluctuate in a similar manner, tracking algal respiration at night (lower pH) and photosynthesis during the day (elevated pH) due to $CO_2$ uptake (day) and release (night). pH is a key factor in the availability of the toxic unionized form of ammonia-N. Thus elevated pH can have a direct impact not only on the aquatic biota, but also on NPDES permit limitations. The degree of the fluctuations for both D.O. and pH are a reflection of the activity of algae, with higher fluctuations corresponding to, amongst other factors, a greater effect of nutrient enrichment, particularly from elevated TP. The D.O. currently data used in the NE Illinois IPS is based on daytime grab samples from which the minimum and maximum values during the warmest period (July-early September) were used as coarse indicators of eutrophication caused by increased algal activity. The aforementioned combined nutrient approaches include the width of the diel D.O. swing based on continuous D.O. data collected with continuous recording instruments. Such data has been collected by IEPA, DRSCW, and MBI on behalf of the other NE Illinois watershed groups, but it has not been incorporated into the current version of the NE Illinois IPS due to its more limited spatial coverage vis a vis the IPS sites. The goal is to develop an integrated database that can be used in lieu of grab sample data and in support of combined nutrient assessments that include chlorophyll a (sestonic and benthic), related chemical parameters (e.g., TKN, TSS, BOD), habitat (QHEI), and fish and macroinvertebrate assemblage performance. While this data is being assessed on a preliminary basis as part of the annual watershed bioassessments (MBI 2018, 2019, 2020a,b, 2021, and 2022) and for the development of Nutrient Assessment Reduction Plans (NARPs)/Nutrient Implementation Plans (NIPs) by the WWTPs via the watershed groups, a broader effort to include continuous D.O. and chlorophyll a data from as many sites as are available across the NE Illinois IPS study area (and beyond if necessary) is presently being undertaken (MBI 2023). In the meantime, the minimum and maximum D.O. thresholds in the IPS framework will serve as proxy indicators as more quantitative thresholds based on continuous D.O. data are developed. The inclusion of benthic and sestonic chlorophyll a data for the NE Illinois IPS region is in line with the recommendations of the Illinois NSAC (2018). In the interim, however, thresholds based on a correspondence of TP, $N0_3$-N, maximum and minimum D.O., and related parameters such as TKN, TSS, VSS, and BOD to nutrient sensitive fish species and macroinvertebrate taxa and the fIBI and mIBI will be used in the NE Illinois IPS framework.

## RESULTS AND DISCUSSION

Since 2006 the identification of stressors linked to biological impairment has relied on a lines-of-evidence approach. This approach utilizes existing water quality criteria, biological response signatures , stressor thresholds developed from an earlier IPS completed in 2010 (Miltner et al. 2010), and stressor thresholds “borrowed” from neighboring states and regions (e.g., Ohio EPA 1999, MBI 2015) and elsewhere (see Table 6 in this Chapter). This current revision of the NE Illinois IPS includes data from a broader geographic area including sites from the five watershed groups and reference sites supplemented by data collected by IEPA/IDNR in adjoining watersheds and counties. For the NE Illinois IPS, a regional dataset consisting of paired biological, chemical, and physical data across seven (7) Illinois Level IV subregions (see Figure 6) was used to derive stressor effect thresholds for 31 water column parameters (Table 8), 31 sediment chemistry parameters (Table 9), and 25 habitat and land use variables (Table 10) that are stratified by the five narrative categories of the fIBI and mIBI. In all, 87 thresholds were derived from a total dataset of 139 water column parameters, 144 sediment parameters, 16 habitat variables, and 39 land use variables, each of which were paired with the biological data at the site level across a total of 640 sites in the NE Illinois IPS study area. As such, the results comprise an Atlas of Stressor Thresholds for the array of stressors that are associated with a gradient of biological conditions in NE Illinois rivers and streams and used herein to derive Restorability scores for impaired sites and Threat and Susceptibility scores for sites that fully attain the General Use criteria for aquatic life. The FIT factor and the regional reference values are included for each variable or parameter in Tables 8-10 and ranked by the strength of the FIT factor from strongest to weakest.

For certain chemical water column parameters, the corresponding water quality criteria are also listed. This was done for the metals parameters because the SSD derived thresholds were much lower than the chronic and acute water quality criteria and because a truer representation of

***Table 8**. Biological effect thresholds and regional reference values for 31 water column parameters, the former derived using Sensitive Species Distributions (SSDs) and relating it back to fish and macroinvertebrate IBI narrative condition categories (Excellent, Good, Fair, Poor, and Very Poor). The variables are ranked by their respective FIT scores from strongest (closest to 0) to weakest. The parameter code, units, most limiting assemblage, and sample size are also shown. The Illinois WQS chronic and acute standards for metals are included (red) at a hardness of 300 mg/L as $CaCO_3$.*

| Parameter Code | Variable Name | Units | Parameter Group | Limiting Assemblage | FIT Score | Sample N | Thresholds by Narrative Condition Category | | | | | Reference Site Values (Median-2X IQR) | Reference Site N |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | Excellent | Good | Fair | Poor | Very Poor | | |
| P665 | Total Phosphorus | mg/L | Nutrients | Fish | 0.04 | 1464 | ≤0.106 | ≤0.277 | >0.277 | >1.002 | >1.726 | 0.088 (0.062-0.115) | 35 |
| P94 | Conductivity | µS/cm | Ionic | Fish | 0.05 | 1464 | ≤739 | ≤1038 | >1038 | >1208 | >1378 | 922 (705-1158) | 40 |
| P70300 | Total Dissolved Solids | mg/L | Ionic | Fish | 0.10 | 1464 | ≤453.8 | ≤558.0 | >558.0 | >651.2 | >744.5 | 614 (512-664) | 28 |
| DO_MIN | Minimum DO | mg/L | Demand | Macros | 0.10 | 985 | >8.0 | ≥6.5 | <5.47 | <4.44 | <3.40 | 8.6 (6.5-9.6) | 29 |
| P625 | Total Kjeldahl Nitrogen | mg/L | Demand | Macros | 0.14 | 985 | ≤1.07 | ≤1.12 | >1.12 | >1.63 | >2.14 | 0.74 (0.30-0.99) | 30 |
| P940 | Chloride, Total | mg/L | Ionic | Fish | 0.17 | 1464 | ≤40.00 | ≤120.00 | >120.0 | >184.9 | >249.8 | 154 (80.3-171.3) | 33 |
| P299 | Mean Dissolved Oxygen | mg/L | Demand | Macros | 0.21 | 985 | >9.42 | ≥9.25 | <9.25 | <6.11 | <3.05 | 8.6 (7.9-9.0) | 40 |
| P310 | BOD (5-Day) | mg/L | Demand | Macros | 0.21 | 985 | ≤1.30 | ≤2.35 | >2.35 | >3.45 | >4.54 | 2 (2.0-2.2) | 27 |
| P610 | Total Ammonia | mg/L | Nutrients | Macros | 0.28 | 985 | ≤0.084 | ≤0.100 | >0.100 | >0.190 | >0.280 | 0.1 (0.10-0.10) | 34 |
| P630 | Nitrate-N | mg/L | Nutrients | Fish | 0.29 | 1464 | ≤3.767 | ≤5.045 | >5.045 | >7.344 | >9.643 | 0.39 (0.29-0.97) | 32 |
| P929 | Sodium, Total | mg/L | Ionic | Fish | 0.29 | 1464 | ≤16275 | ≤4500 | >45000 | >79056 | >113112 | 14200 (10375-22500 | 21 |
| P530 | Total Suspended Solids | mg/L | Demand | Fish | 0.32 | 1464 | ≤17.50 | ≤31.60 | >31.60 | >35.15 | >38.69 | 9.2 (5.4-20.3) | 33 |
| P615 | Nitrite-N | mg/L | Nutrients | Macros | 0.41 | 985 | ≤0.014 | ≤0.040 | >0.040 | >0.068 | >0.096 | 0.01 (0.01-0.01) | 27 |
| DO_MAX | Maximum DO | mg/L | Demand | Macros | 0.94 | 985 | ≤10.36 | ≤12.21 | >12.21 | >14.24 | >16.28 | 8.74 (8.21-9.45) | 29 |
| P82078 | Turbidity | NTU | Demand | Macros | 2.61 | 985 | -- | ≤19.3 | >19.3 | >25.9 | >32.5 | 11.0 (4.5-24.5) | 7 |
| P549 | Volatile Suspended Solids | mg/L | Demand | Fish | 2.81 | 1464 | ≤5.000 | ≤7.769 | >7.769 | >9.825 | >11.88 | 6.0 (4.8-7.4) | 5 |
| P945 | Sulfate, Total | mg/L | Ionic | Macros | 6.49 | 985 | ≤58.27 | ≤73.1 | >73.10 | >83.45 | >93.81 | 74.6 (61.8-81.8) | 4 |
| P937 | Potassium, Total | mg/L | Ionic | Macros | 10.13 | 985 | ≤3158 | ≤6300 | >6300 | >7718 | >9129 | 2400 (1574-2817) | 21 |
| P916 | Calcium, Total | mg/L | Ionic | Fish | Unimodal | 1464 | ≤84425 | ≤86067 | >86067 | >86313 | >86559 | 54,000 (80-74,250) | 21 |
| ***Metals and Toxics*** | | | | | | | | | | | | | |
| P1092 | Zinc, Total | µg/L | Metal_Tox | Fish | 0.13 | 1464 | ≤7.47 | ≤9.78 [CS: 55.5] | >9.78 | >11.00 | >12.22 [309.7] | 2.0 (2.0-7.0) | 23 |
| P1027 | Cadmium, Total | µg/L | Metal_Tox | Fish | 0.93 | 1464 | ≤0.937 | ≤0.974 [CS: 2.70] | >0.974 | >0.983 | >0.991 [33.63] | <MDL (0.17) | 23 |
| P1042 | Copper, Total | µg/L | Metal_Tox | Fish | 1.75 | 1464 | -- | ≤4.480 [CS: 18.65 ] | >4.480 | >4.969 | >5.458 [AS: 30.1] | 2.00 (1.96-4.15) | 22 |
| P1051 | Lead, Total | µg/L | Metal_Tox | Macros | 2.11 | 985 | ≤2.851 | ≤3.335 [CS; 18.0] | >3.335 | >3.884 | >4.434 [AS: 343] | 0.24 (0.20-0.57) | 23 |
| P1082 | Strontium | µg/L | Metal_Tox | Fish | 2.69 | 1464 | ≤169.1 | ≤190.8 | >190.8 | >280.4 | >370.1 | 150 (135-181) | 21 |
| P1055 | Manganese, Total | µg/L | Metal_Tox | Macros | 2.74 | 985 | ≤53.71 | ≤77.03 [CS: 3319] | >77.03 | >107.1 | >137.2 [AS: 7808] | 32.0 (24.1-38.2) | 23 |
| P1067 | Nickel, Total | µg/L | Metal_Tox | Macros | 3.26 | 985 | -- | ≤3.470 [CS: 103.6] | >3.470 | >9.585 | >15.70 [AS: 932] | 5.0 (1.5-21) | 14 |
| P1105 | Aluminum, Total | µg/L | Metal_Tox | Fish | 4.54 | 1464 | ≤310.0 | ≤393.3 | >393.3 | >560.2 | >727.0 | 200 (128-449) | 21 |
| P1007 | Barium, Total | µg/L | Metal_Tox | Fish | 4.77 | 1464 | ≤74.1 | <84.88 | >84.88 | >101.8 | >118.6 | 56.3 (44.3-64.7) | 21 |
| P720 | Cyanide, Total | µg/L | Metal_Tox | Macros | 5.17 | 985 | ≤8 | ≤10 [CS: 5.2] | >10 | >10 | >10 [AS: 22] | 3 (2-10) | 6 |
| P1002 | Arsenic | µg/L | Metal_Tox | Macros | 9.19 | 985 | -- | ≤3.455 [CS: 190] | >3.455 | >5.029 | >6.603 [AS: 360] | Insufficient Data | |
| P1034 | Chromium, Total | µg/L | Metal_Tox | Fish | 10.17 | 1464 | ≤1.398 | ≤1.540 [CS: 167] | >1.540 | >2.682 | >3.824 [AS: 3503] | 1.73 (1.30-2.00) | 6 |

CS - Illinois WQS chronic standard equated to Good; AS - Illinois WQS acute standard equated to Very Poor.

***Table 9.*** *Biological effect thresholds and regional reference values for 31 sediment chemistry parameters, the former derived using Sensitive Species Distributions (SSDs) and relating it back to fish and macroinvertebrate IBI narrative condition categories (Excellent, Good, Fair, Poor, and Very Poor). The variables are ranked by their respective FIT scores from strongest (closest to 0) to weakest. The parameter code, units, most limiting assemblage, and sample size are also shown. The lowest and threshold and probable and severe thresholds from the Canadian Council of Ministers of the Environment (CCME 1993) and MacDonald et al. (2000) and metals values from Short (1998) are included for comparison under Literature Thresholds.*

| Parameter Code | Variable Name | Units | Parameter Group | Limiting Assemblage | FIT Score | Sample N | Thresholds by Narrative Condition Category | | | | | Literature Thresholds | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | Excellent | Good | Fair | Poor | Very Poor | TEC/LEL | PEC/PEL | Short | Source |
| P1093 | Zinc | mg/kg | Metal_Tox | Macros | 2.22 | 985 | ≤75.00 | ≤100.0 | >100.0 | >133.9 | >167.8 | 121 | 459 | 170.0 | MacDonald |
| P34524 | Benzo(g,h,i)perylene | µg/kg | PAH | Macros | 2.32 | 985 | -- | ≤335.0 | >335.0 | >792.1 | >1249 | 170 | 320 | | MacDonald |
| P34406 | Indeno(1,2,3-cd)pyrene | µg/kg | PAH | Macros | 2.41 | 985 | -- | ≤260.5 | >260.5 | >623.3 | >986.2 | 200 | 3200 | | MacDonald |
| P1043 | Copper | mg/kg | Metal_Tox | Macros | 2.42 | 985 | ≤19.00 | ≤29.78 | >29.78 | >40.45 | >51.12 | 31.6 | 149 | 37.0 | MacDonald |
| P34233 | Benzo(b)fluoranthene | µg/kg | PAH | Macros | 2.51 | 985 | -- | ≤520.8 | >520.8 | >1437 | >2354 | 240 | 13400 | | MacDonald |
| P1068 | Nickel | mg/kg | Metal_Tox | Macros | 2.67 | 985 | -- | ≤19.50 | >19.50 | >22.52 | >25.53 | 22.7 | 48.6 | 26.0 | MacDonald |
| P34250 | Benzo(a)pyrene | µg/kg | PAH | Macros | 2.85 | 985 | -- | ≤230.0 | >230.0 | >798.3 | >1367 | 150 | 1450 | | MacDonald |
| P34472 | Pyrene | µg/kg | PAH | Macros | 2.85 | 985 | -- | ≤393.0 | >393.0 | >1570 | >2747 | 195 | 1520 | | MacDonald |
| P1052 | Lead | mg/kg | Metal_Tox | Macros | 3.01 | 985 | ≤15.50 | ≤24.80 | >24.80 | >33.04 | >41.27 | 35.8 | 128 | 60.0 | MacDonald |
| P34529 | Benzo[a]anthracene | µg/kg | PAH | Macros | 3.48 | 985 | -- | ≤239.0 | >239.0 | >699.4 | >1160 | 108 | 1050 | | MacDonald |
| P34323 | Chrysene | µg/kg | PAH | Macros | 3.51 | 985 | -- | ≤266.0 | >266.0 | >958.3 | >1651 | 166 | 1290 | | MacDonald |
| P34379 | Fluoranthene | µg/kg | PAH | Macros | 3.91 | 985 | -- | ≤774.0 | >774.0 | >2432 | >4091 | 423 | 2230 | | MacDonald |
| P1083 | Strontium | mg/kg | Metal_Tox | Macros | 4.44 | 985 | -- | ≤81.80 | >81.80 | >106.8 | >131.9 | None | None | | |
| P34559 | Dibenz(a,h)anthracene | µg/kg | PAH | Macros | 4.57 | 985 | -- | ≤101.0 | >101.0 | >167.3 | >233.7 | 33 | 135 | | MacDonald |
| P34223 | Anthracene | µg/kg | PAH | Macros | 5.10 | 985 | -- | ≤78.00 | >78.00 | >119.9 | >161.8 | 46.9 | 245 | | CCME |
| P34464 | Phenanthrene | µg/kg | PAH | Macros | 5.10 | 985 | -- | ≤243.5 | >243.5 | >803.3 | >1363 | 204 | 1170 | | MacDonald |
| P1003 | Arsenic | mg/kg | Metal_Tox | Macros | 6.21 | 985 | -- | ≤8.65 | >8.65 | >15.82 | >23.67 | 9.79 | 33 | 7.2 | MacDonald |
| P1029 | Chromium | mg/kg | Metal_Tox | Macros | 6.29 | 985 | ≤20.53 | ≤23.30 | >23.30 | >26.22 | >29.15 | 43.4 | 111 | 37.00 | MacDonald |
| P1053 | Manganese | mg/kg | Metal_Tox | Macros | 7.08 | 985 | ≤841.0 | ≤845.5 | >845.5 | >996.8 | >1148 | 460 | 1100 | 1100 | MacDonald |
| P1078 | Silver | mg/kg | Metal_Tox | Macros | 7.11 | 985 | -- | ≤0.483 | >0.483 | >1.261 | >2.039 | 1.6 | 2.2 | | MacDonald |
| P1108 | Aluminum | mg/kg | Metal_Tox | Macros | 8.26 | 985 | -- | ≤6480 | >6480 | >8272 | >10064 | | | | |
| P1008 | Barium | mg/kg | Metal_Tox | Macros | 8.88 | 985 | -- | ≤141.0 | >141.0 | >150.3 | >168.7 | | | 145 | |
| P1028 | Cadmium | mg/kg | Metal_Tox | Macros | 11.00 | 985 | -- | ≤0.745 | >0.745 | >1.354 | >1.963 | 0.99 | 4.98 | 2.000 | MacDonald |
| P1013 | Beryllium | mg/kg | Metal_Tox | Macros | ND[a] | 985 | -- | ≤0.411 | >0.411 | >0.496 | >0.581 | | | | |
| P1103 | Tin | mg/kg | Metal_Tox | Macros | ND[a] | 985 | -- | ≤8.86 | >11.00 | >16.73 | >24.60 | | | | |
| P34203 | Acenaphthylene | µg/kg | PAH | Macros | ND[a] | 985 | -- | ≤86.38 | >86.38 | >103.6 | >120.9 | 5.87 | 128 | | CCME |
| P34208 | Acenaphthene | µg/kg | PAH | Macros | ND[a] | 985 | -- | ≤84.25 | >84.25 | >104.8 | >125.3 | 6.71 | 88.9 | | CCME |
| P34262 | Delta-BHC | µg/kg | PAH | Macros | ND[a] | 985 | -- | ≤2.098 | >2.098 | >6.19 | >10.28 | | | | |
| P34384 | Fluorene | µg/kg | PAH | Macros | ND[a] | 985 | -- | ≤84.25 | >84.25 | >104.8 | >125.3 | 77.4 | 536 | | MacDonald |
| P34445 | Naphthalene | µg/kg | PAH | Macros | ND[a] | 985 | -- | ≤86.38 | >86.38 | >103.6 | >120.9 | 34.6 | 391 | | CCME |

[a] - Not determined (ND) due to a high number of non-detects

MacDonald - MacDonald, D. D., C. G. Ingersoll, and T. A. Berger. 2000. Development and Evaluation of Consensus-Based Sediment Quality Guidelines for Freshwater Ecosystems. Arch. Environ. Contam. Toxicol. 39, 20–31.

CCME - Canadian Council of Ministers of the Environment (CCME). 1999. Canadian sediment quality guidelines for the protection of aquatic life. Canadian environmental quality guidelines, 1999, Canadian Council of Ministers of the Environment, Winnipeg, MB.

***Table 10.*** *Biological effect thresholds and regional reference values for 25 habitat and land use variables, the former derived using Sensitive Species Distributions (SSDs) and relating it back to fish and macroinvertebrate IBI narrative condition categories (Excellent, Good, Fair, Poor, and Very Poor). The variables are ranked by their respective FIT scores from strongest (closest to 0) to weakest. The parameter code, units, most limiting assemblage, and sample size are also shown.*

| Parameter Code | Variable Name | Units | Parameter Group | Limiting Assemblage | FIT Score | Sample N | Thresholds by Narrative Condition Category | | | | | Reference Site Values (Median - 2X IQR) | Reference Site N |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | Excellent | Good | Fair | Poor | Very Poor | | |
| EMBEDDED | Embeddedness Score | QHEI Units | Habitat | Fish | 0.03 | 1393 | ≤1.3 | ≤1.6 | >1.6 | >2.4 | >3.2 | 2 (2-2) | 29 |
| Urban | Urban (Ust. WS) | Wtd. % | Land Use | Fish | 0.03 | 2657 | ≤8.8 | ≤45.0 | >45.0 | >63.2 | >81.3 | 8.7 (3.0-9.5) | 48 |
| QHEI | QHEI Score | QHEI Units | Habitat | Fish | 0.04 | 1393 | ≥84.5 | ≥75.9 | <75.9 | <50.1 | <25.0 | 84 (76-90) | 34 |
| SUBSTRAT | Substrate Score | QHEI Units | Habitat | Fish | 0.04 | 1393 | ≥16.0 | ≤15.0 | <15.0 | <9.9 | <5.0 | 8 (7-9) | 33 |
| WWH_ATTR | Good Habitat Attributes | Number | Habitat | Fish | 0.04 | 1393 | ≥9 | ≥8 | <8 | <5 | <2 | 16 (15-17) | 34 |
| Imperv | Impervious (30 m) | Wtd. % | Land Use | Fish | 0.04 | 2657 | ≤18.3 | ≤30.5 | >30.5 | >53.4 | >76.4 | 2.1 (0.0-14.7) | 48 |
| Imperv | Impervious (30 m Clipped) | Wtd. % | Land Use | Fish | 0.04 | 2657 | ≤13.4 | ≤26.7 | >26.7 | >50.9 | >75.1 | 2.1 (0.0-6.1) | 48 |
| CHANNEL | Channel Score | QHEI Units | Habitat | Fish | 0.07 | 1393 | ≥16.8 | ≥14.0 | <14.0 | <9.2 | <4.6 | 16 (13-19) | 34 |
| COVER | Cover Score | QHEI Units | Habitat | Fish | 0.07 | 1393 | ≥16.0 | ≥14.0 | <14.0 | <9.2 | <4.6 | 16 (16-17) | 34 |
| SILTCOVE | Silt Cover Score | QHEI Units | Habitat | Fish | 0.07 | 1393 | -- | ≤2.0 | >2.0 | >2.7 | >3.33 | 2 (2-3) | 29 |
| Develop | Developed (Ust. WS) | Wtd. % | Land Use | Fish | 0.07 | 2657 | ≤9.1 | <45.6 | >45.6 | >63.6 | >81.5 | 9.1 (2.9-9.6) | 48 |
| RIPARIAN | Riparian Score | QHEI Units | Habitat | Fish | 0.10 | 1393 | -- | ≥6.0 | <6.0 | <4.0 | <2.0 | 7.0 (6.0-9.5) | 34 |
| Imperv | Impervious (Ust. WS) | Wtd. % | Land Use | Macros | 0.10 | 3096 | ≤5.6 | ≤13.2 | >13.2 | >41.8 | >70.5 | 5.2 (2.1-5.4) | 48 |
| DEPTH | Depth Score | QHEI Units | Habitat | Fish | 0.11 | 1393 | -- | ≥10.0 | <10.0 | <6.6 | <3.3 | 10 (9-11) | 33 |
| MWH_ATTR | Poor Habitat Attributes | Number | Habitat | Fish | 0.12 | 1393 | <u>0</u> | 1 | >1 | >3 | >6 | 2 (1-5) | 20 |
| HYD_QHEI | Hydro-QHEI | QHEI Units | Habitat | Fish | 0.13 | 1393 | ≥17.0 | ≤19.5 | <19.5 | <12.9 | <6.4 | 20 (14-22) | 33 |
| CURRENT | Current Score | QHEI Units | Habitat | Fish | 0.14 | 1393 | -- | ≥7.0 | <7.0 | <4.6 | <2.3 | 11 (5.8-11.0) | 33 |
| POOL | Pool Score | QHEI Units | Habitat | Fish | 0.15 | 1393 | ≥11.25 | <11.25 | <10.0 | <6.6 | <3.3 | 11.5 (10-12) | 34 |
| Heavurb | Heavy Urban (Ust. WS) | Wtd. % | Land Use | Macros | 0.17 | 3096 | ≤7.7 | <29.3 | >29.3 | >52.6 | >76.0 | 5.5 (1.1-6.0) | 48 |
| RIFFLE | Riff< Score | QHEI Units | Habitat | Fish | 0.27 | 1393 | ≥5.88 | ≥5.75 | <5.75 | <3.9 | <1.9 | 6 (5-7) | 34 |
| GRADIENT | Gradient Score | QHEI Units | Habitat | Fish | 0.31 | 1393 | -- | ≥10.0 | <10.0 | <6.6 | <3.3 | 10 (10-10) | 34 |
| Ag | Agricultural (Ust. WS) | Wtd. % | Land Use | Macros | 4.82 | 3096 | ≥87.1 | ≥62.1 | <62.1 | <41.4 | <20.5 | 83.9 (11.7-85.4) | 48 |
| GRADIENT | Gradient (ft/mi) | feet/mile | Habitat | Fish | 12.20 | 1393 | ≥8.8 | ≥4.3 | <4.3 | <2.8 | <1.4 | 8.6 (4.9-11.3) | 34 |
| Ag | Agricultural (30 m) | Wtd. % | Land Use | Macros | 16.66 | 3096 | ≤3.4 | ≤10.1 | >10.1 | >29.7 | >59.97 | 0.0 (0.0-0.4) | 48 |

poor and very poor values were simply not available given the dearth of truly toxic values among the data for these parameters. In this case the Illinois WQS would be used in lieu of the SSD derived thresholds. Historical data that would likely have higher concentrations of metals and other parameters is spatially sparse, but it could provide the basis for simulating more complete stressor gradients. The historical database utilized by Capmourteres et al. (2018; see Table 3) could be a partial basis for such future improvements to the IPS thresholds. For the sediment chemical parameters, the consensus thresholds (Low, Threshold, Severe, and Probable Effect Levels) of Persaud et al. (1993), CCME (1999), and MacDonald et al. (2000) and sediment metals benchmarks of Short (1998) based on Illinois data were also included for reference and usage. The lack of sediment chemistry data from enough NE Illinois Excellent sites precluded the derivation of an Excellent threshold for all but five (5) sediment parameters.

### Field-Derived Species and Taxa Stressor Thresholds

The field derived species and taxa based SSDs supported establishing a range of aquatic life use goals, the acceptable minimum of which is reflective of least impacted reference conditions. This approach was used by IEPA to derive the Illinois General Use fIBI and mIBI thresholds for 305[b] reporting and 303[d] listing purposes (IEPA 2018, 2022). General Use thresholds for each chemical, habitat, and land use variable were then derived as the 75th percentile at sites meeting the fIBI or mIBI General Use threshold that also had >25th percentile of stressor specific sensitive fish species or macroinvertebrate taxa present. Previous studies have used quantile regression curves (sensitive species/taxa vs. the stressor and sensitive species/taxa vs. an IBI value) to derive stressor-specific threshold values, but the distribution of sites in the NE Illinois IPS database was strongly skewed towards the fair, poor and very poor narratives and so lacked sufficient upper anchors in Good and Excellent condition. This introduced unintended variability in the quantile regression slopes and intercept points normally used to derive thresholds. Because of this we used a "hybrid" approach that employs a combination of quantile regression and visual examination to examine the form of the relationship between stressors and species/taxa richness. Using assemblage based IBI measures *and* baseline measures of stressor-specific sensitive species/taxa ensures that the thresholds are protective of the narrative category for that stressor as follows:

1) Biological performance consistent with the IEPA narrative categories of Excellent, Good (minimally meets the Illinois General Use for aquatic life), Fair, Poor, and Very Poor; and,
2) Sites above the stressor-specific $25^{th}$ percentile threshold for stressor-specific sensitive fish species or macroinvertebrate taxa, whichever was the most limiting, at attaining sites (good and excellent).

This is similar to the approach taken by Bryce et al. (2010) who also did not rely solely on an IBI threshold to set thresholds for bedded sediments:

> *". . . although IBIs typically incorporate sensitive-taxon metrics that show strong responses to sediment, they also contain metrics that capture assemblage response to other stressors. As a result, choosing an IBI score considered good (e.g., 80) and*

*matching it with its corresponding %fine sediment on a quantile regression plot of IBI vs %fines does not ensure a limit (or criterion value) that will be protective of sediment-intolerant species within the aquatic assemblage. Rather, we show that a method of sediment criteria development that focuses on identifying minimum-effect sediment levels is more likely to be protective of sediment-sensitive aquatic species."*

Thus, and herein for the IPS, groupings of stressor-specific sensitive species and taxa were used to derive protective effect thresholds as the basis for the outputs of the IPS framework.

The stressor effect thresholds were based on the concentrations of levels of stressors at attaining sites that also have greater than or equal to the 25$^{th}$ percentile of sensitive species at these sites. When stressor levels at attaining sites were examined, there was a distinct separation in stressor concentrations for attaining sites that were above or below the 25$^{th}$ percentile of stressor-sensitive species (see Figure 19). This suggests that a threshold derived using data from all sites could result in conditions that may adversely affect a stressor-specific sensitive species or taxon. High numbers of stressor-specific species/taxa are likely to be present at Excellent performing sites, hence, the derivation of an Excellent threshold to serve as a level of protection against future degradation is warranted. Low numbers or an absence of stressor-specific sensitive species/taxa at sites that minimally attain the General Use biocriteria and the Good narrative are considered threatened.

The following section discusses the stressor threshold results by key parameter groupings. The NE Illinois dataset was adequate to develop stressor thresholds across the five narrative condition categories for most of the common habitat, land use, chemical water column, and some sediment chemistry variables and parameters. However, for some parameters the inconsistency in the availability of data at sufficient sites resulted in some minor differences in deriving thresholds primarily the lack of sufficient data at Excellent performing sites. These occur mainly in the outlying parts of the NE Illinois IPS study area. Sediment chemical data, for example, was lacking at most of the outlying and higher quality sites which made it difficult to develop an Excellent threshold for heavy metals and most PAH compounds. Regional reference values were also derived based on a mix of watershed group and IEPA reference sites, but here again the data was inconsistent for selected parameters, especially sediment chemistry. The various plots of the stressor sensitive species/taxa and IBI relationships are shown in Figure 19 for the QHEI score and selected QHEI metrics and attributes. The remaining plots for all other parameters appear in Appendix C.

**Habitat Parameters**

Habitat as portrayed herein by the QHEI and its metrics and attributes, is one of the most prevalent and limiting stressors to aquatic life in Midwest rivers and streams. Thresholds for the QHEI score and 15 QHEI attributes that had significant relationships in the stressor analyses are listed in Table 10 for each of the five narrative condition categories. The QHEI score and the 15 attributes had the most limiting relationships with fish assemblage measures. The QHEI score showed a strong threshold relationship with QHEI-sensitive fish species (Figure 18, top left) and

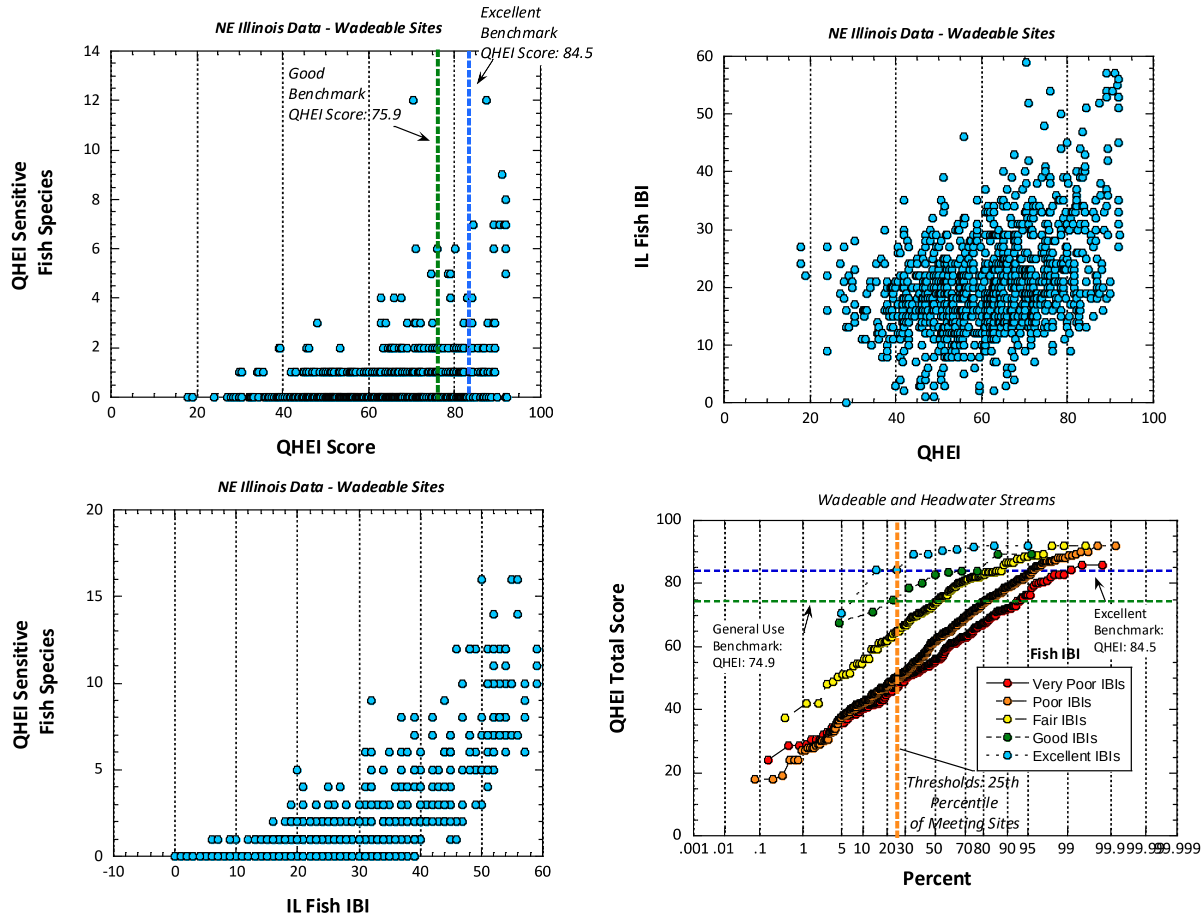
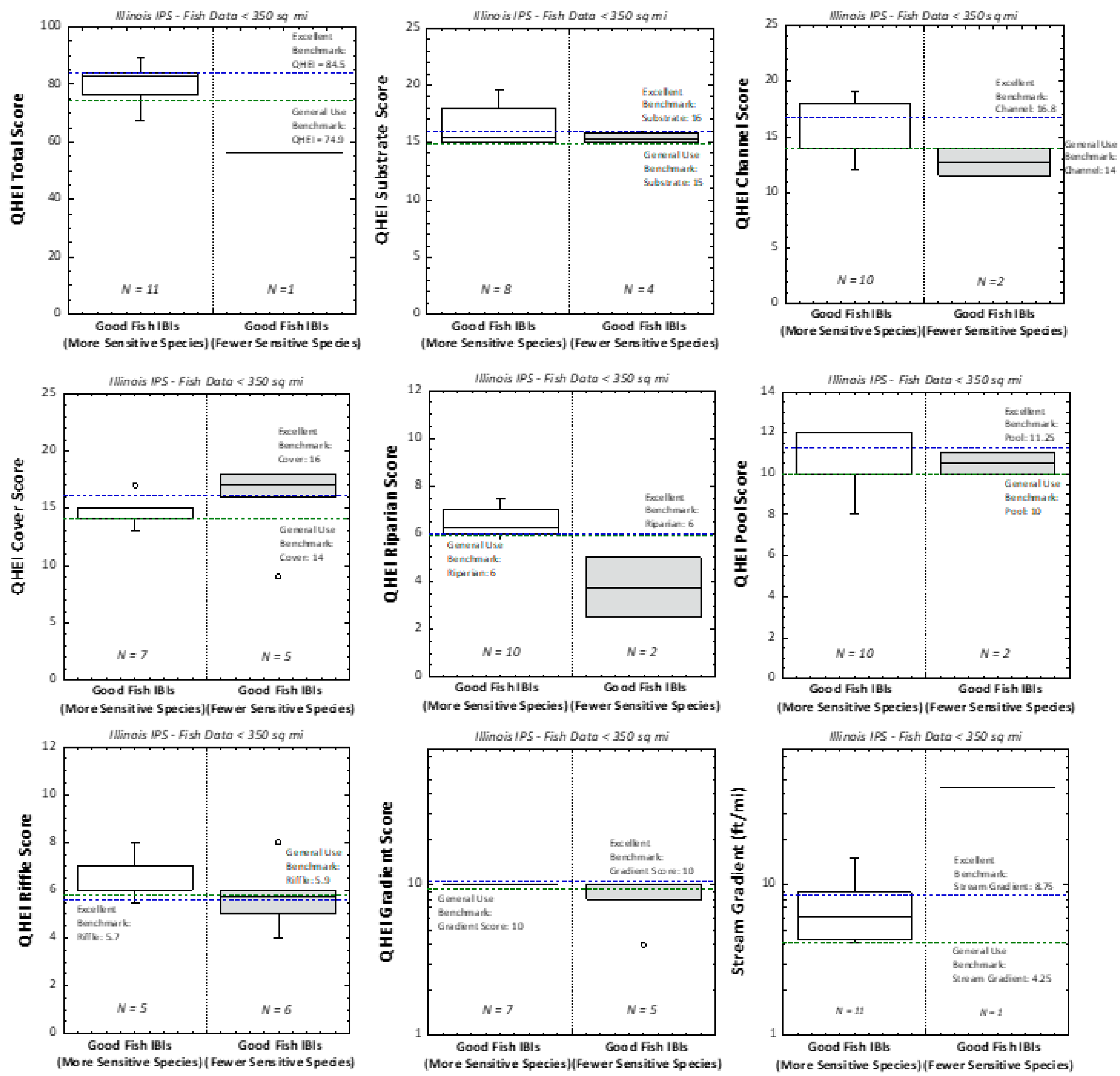



***Figure 18**. Plots supporting derivation of QHEI total score thresholds for wadeable streams in NE Illinois including scatter plots of QHEI vs QHEI sensitive fish species (top left), Fish IBI vs. QHEI sensitive fish species (bottom left), QHEI vs. Fish IBI (top right) and a probability plot of QHEI values by narrative ranges of the IBI. Data from Illinois IPS study area sites in NE Ohio (see text).*

with the fIBI (Figure 18, bottom left). There is also a distinct threshold relationship between the QHEI score and the fIBI (Figure 18, top right), although there is more scatter in that relationship than with the QHEI sensitive fish species (Figure 18, top left). As discussed earlier, the fIBI is designed to respond to a wide range of stressors and thus we derived the QHEI sensitive species measure to provide more clarity in the response of biota to habitat.

Figure 19 illustrates the differences in 11 of the habitat metrics and attributes between sites with good fIBIs and >25$^{th}$ percentile of QHEI-sensitive fish species and sites with good IBIs and <25$^{th}$ percentile value of QHEI-sensitive fish species. The QHEI score and most metric scores are higher at the good sites with more QHEI sensitive fish species than at sites with fewer than the 25$^{th}$ percentile of QHEI sensitive species (Figure 19). Those sites with fewer than the 25$^{th}$ percentile number of sensitive fish species will have an increased Threat score. This provides evidence that having the number of QHEI sensitive fish species below the Good threshold are either providing marginal habitat and/or are influenced by being in a HUC12 watershed with



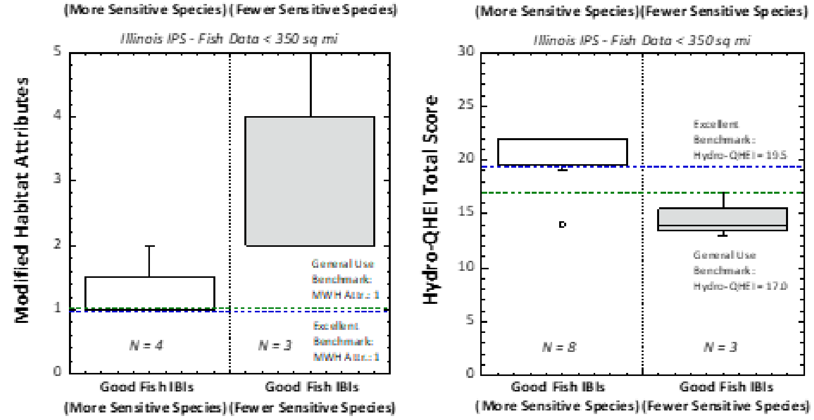




***Figure 19**. Box and whisker plot for the QHEI and key metrics comparing sites with good IBI scores and more sensitive fish species with sites good IBI and fewer sensitive fish species (< 25th percentile for attaining sites).*

widespread marginal habitat and are likely vulnerable to further habitat degradation. Given the influence of watershed and reach scale on habitat (Rankin 1995), the identification of restoration or protection options for sites should consider both the reach and watershed scale Threat scores to inform the selection of appropriate protection and restoration strategies.

**Land Use Variables**

Land use data has been shown to be strongly correlated with biological assemblage quality in urban and suburban landscapes (Walsh et al. 2005). There have been a number of efforts to derive apparent relationships between the degree of urbanization in the form of impervious land cover and related land use measures with biological indicators (Schueler 2004, Schueler et al. 2009). For the NE Illinois IPS framework we evaluated a suite of land use measures derived by DRSCW and DRWW for sites across the NE Illinois IPS study area. Chicago Agency for Planning (CMAP) and National Land Cover (NLC) land use datasets were used to generate land use categories at the watershed, 500 meter buffer, and 30 meter spatial buffer scale. Polygons clipped to the immediate watershed and unclipped at the watershed-scale were generated. Land use categories were stratified into broader categories of agricultural, developed, forested, natural, wetland, grassland, and heavy urban land uses and the proportion of impervious cover was calculated. While other combinations and measures are possible, these provided an initial set of the most widely recognized land use variables.

Land use is considered to be source level data that coincides with certain proximate stressors that are most likely the agents or causes that are limiting to aquatic assemblages. Certain land uses can be strongly correlated with stressor conditions such as altered flow regimes, degraded habitat, and degraded chemical water quality. The results of the analyses allowed for more accurate weighting of the variables that comprise the Restorability, Susceptibility, and Threat scores.

Biological responses related to eight (8) land uses at the watershed scale was the strongest for the developed land uses (Table 10). The most sensitive assemblage was split evenly between fish and macroinvertebrates at four variables each. These variables represented land cover types in the entire watershed upstream of a sampling point and the clipped polygons immediately upstream of a site. Land use categories such as agricultural, forested, natural, and wetland land cover showed little relationship with the aquatic assemblage data. The results for agricultural land uses was included in Table 10 as an illustration of the unreliability of that factor alone to meaningfully correspond to assemblage condition and with some of the highest FIT values of any chemical, habitat, physical, or sediment chemistry variables.

The strongest relationships were with heavy urban land use and the various impervious cover (IC) measures (Table 10). Prior predictions of IC models showing adverse effects at as low as 10% IC (Schueler 2003; Schueler et al. 2009) is consistent with the results shown here. The IC threshold for the Excellent narrative category for NE Illinois is 5.6 % (Table 10). The Good narrative (General Use) threshold of 13.2% is also below where several IC models predict impairment of minimum CWA Section 101(a)(2) goals (Yoder et al. 1999, 2000; Miltner et al.

2003). At 29.3% the heavy urban Good threshold is slightly above the commonly cited 25% urban land use for the CWA goal (Schueler 2003). These results also reveal clear differences between Excellent and Good (General Use) streams and rivers, the latter threshold at 7.7% heavy urban. Plots of the key land use variables for which thresholds were derived are included in Appendix B.

The lack of a clear threshold for forested or natural land uses that would act to insulate against the adverse effects of impervious cover (IC) could be due to the large proportion of sites that at a watershed scale exceeded the 5.6% Excellent and 13.2% Good thresholds of IC in the upstream watershed. In a review of papers related to IC, Schueler et al. (2009) reported that at *"a certain point [15% urban land as identified by Roy et al. (2006) or 10% IC as identified by Goetz et al. (2003)], the degradation caused by upland storm-water runoff shortcutting the buffer overwhelms the more localized benefits of riparian canopy cover."* Conducting additional analyses that more closely examine sites that are below these thresholds would reveal if the spatial buffer land use measures could potentially offset adverse impacts in watersheds with heavy urban and IC greater than the Excellent and Good thresholds.

Figure 20 represents the average land use composition of watershed survey sites contrasted with reference sites at the entire watershed scale (left) and within a 30 meter spatial buffer upstream of each site (right). Median and mean land use data is also summarized in Table 10 for each land use type. At the watershed scale, reference sites had a higher proportion of agricultural land use (average >70%) whereas non-reference watershed survey sites had predominately developed and urban land uses. Both settings had, on average, similar amounts

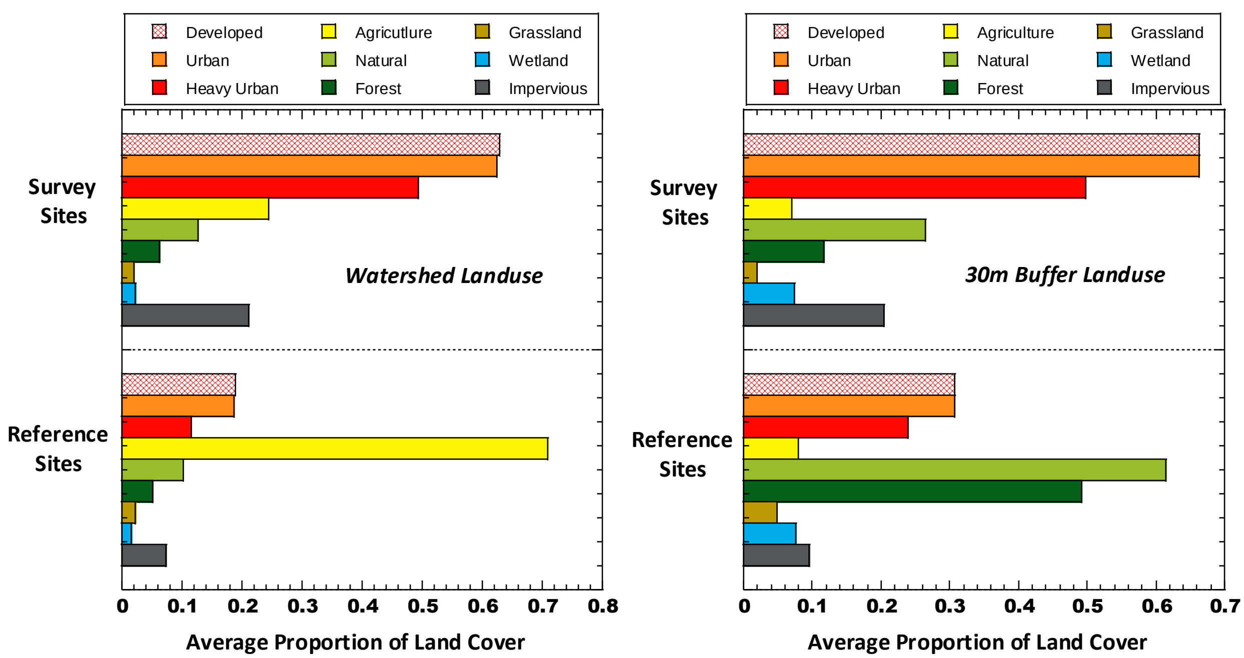



***Figure 20***. *Watershed land use (left) and 30 meter buffer upstream land use (right) at watershed survey sites (top) and reference sites (bottom) for sites with land use data available.*

of forested and natural land uses. Illinois has lost ~90% of its original wetland acres, and wetlands made up only a small percentage of watershed wide land uses for both the watershed survey and reference sites. There was a major difference in the spatial buffer (30 meter width) land uses between the reference sites and survey sites with more forest and natural land uses at reference sites and more developed land uses in spatial buffers at watershed survey sites.

Although watershed-wide land uses are correlated with aquatic life condition, there is evidence that land use in the immediate stream spatial buffer could insulate against some of the otherwise assumed adverse impacts. The land use variables in Table 10 include land use within the 30 and 500 meter spatial buffers in the entire watershed and clipped to the area immediately upstream from a site. These tended to show a higher proportion of urban land use measures corresponding to Excellent and Good conditions compared to the measures of the entire watershed with a mix of the fish and macroinvertebrates being the most limiting assemblages.

The land use in the clipped 30 meter spatial buffer refers to the area immediately upstream and draining to of a sampling site compared to the 30 meter spatial buffer of the entire surrounding area. For the clipped impervious cover (IC) variable the fish assemblage was only somewhat more sensitive than the macroinvertebrate assemblage. The limitation of natural buffers to mitigate the effects of developed land uses when IC increases beyond 10-25% needs to be determined. While local stream conditions may be more challenged at a higher IC, the effects of IC are also exported downstream. Although Schueler et al. (2009) suggests that conditions within streams with >50% IC may be severely limited (termed an urban drainage), local buffer restoration could benefit downstream reaches where tributary watersheds have lower IC. The initial IPS (Miltner et al. 2010) established that *". . . within the domain of measured riparian scores and mIBI scores, based on the standardized regression coefficient (from the SEM model in the DRSCW IPS) each 1 point increase in the riparian score gains ~1.3 mIBI points, that is if the riparian score is increased from 5 to 10 (i.e., from ~25 m to 50 m), the gain, on average, would be about 6.5 mIBI points. That's a fairly substantial gain in a system where every incremental improvement is beneficial."* This illustrates that the limiting effect of IC on incremental improvement and eventual General Use attainment is not well enough understood to simply declare watersheds as "urban limited" on the basis of IC alone.

**Water Column Variables**

These are water quality measures based on periodic (i.e., weekly to monthly) grab water samples collected from the water column and within a seasonal summer-early fall index period. Each are parameters that are measured as laboratory analytes or with hand held meters. A total of 139 parameters were analyzed, but not all were collected at every sampling site nor at the same frequency which affected the ability to derive thresholds and regional reference values for each. However, a core set of parameters that are broadly grouped into four parameter categories (ionic strength, demand, nutrients, and metals/toxics) were collected consistently enough to support the derivation of thresholds for 31 parameters (Table 8). The FIT score for each variable is also included as are the reference sites[10] values where that data was sufficient to calculate a median and the interquartile (IQR) range. All of the threshold derivation plots and results appear in Appendix C.

### *Ionic Strength Parameters*

Ionic strength parameters are primarily in the form of dissolved ions and are produced by point sources and urban runoff, the latter being particularly related to the application of deicing salt during the winter. For the NE Illinois IPS this category included total dissolved solids (TDS), chloride, sulfate, and conductivity. Of these parameters, sulfate, chloride, and total dissolved solids each have numerical criteria in the Illinois WQS (see Table 6).

*Chloride*

Numerous studies have identified strong relationships between summer concentrations of chlorides in rivers and streams and winter application of salt products (Kaushal et al. 2005, 2018; Corsi et al. 2010). This observation is widespread across the continent and has been termed the “freshwater salinization syndrome” (Kaushal et al. 2018). Chloride and other dissolved ions used in winter deicing are not completely exported out of watersheds during winter storm events, a substantial fraction accumulates in the watershed soils of a near stream ground-water riparian areas and on infrastructure (stormwater ponds and application surfaces).A result is elevated summer concentrations to the extent that loadings of chloride and allied measures such as specific conductance and dissolved solids can be used to predict acute events during winter (Trowbridge et al. 2010). Thus, the thresholds in Table 8 are residual concentrations that are not representative of maximum instream concentrations, but rather are the residual concentrations . These are lower than the values that occur in runoff during winter months. As such, the effect thresholds derived from summer-fall chloride levels are representative of the resident biota that are exposed to higher concentrations during the winter months.

The threshold derivation plots for chloride were derived with all data for chloride values in the NE Illinois IPS study area for streams and rivers draining <350 square miles. The fish assemblage was the more limiting of the two assemblages with an Excellent threshold of 40 mg/L and a Good threshold of 120 mg/L. The latter are very close to the threshold derived by the previous IPS (Miltner et al. 2010) and more recently by Miltner (2021) for Ohio streams, but is well below the EPA national criterion of 230 mg/L and the current Illinois WQS acute value of 500 mg/L. The Illinois WQS is one of the more outdated criteria in the Illinois WQS. The IPS thresholds are from summer-fall normal flow values that reflect the residual fraction of chloride loadings that are delivered during the cold weather months and primarily from the application of deicing chemicals containing salt. The above cited studies have shown that annual chloride concentrations are increasing by 1 mg/L per year in the northern tier of U.S. states. The NE Illinois IPS study area results are an indication of the residual concentration of chloride that has built up over time in near surface ground water, the riparian zone, and soils in the watershed. It

[10] Reference sites are a mix of DRSCW and IEPA least impacted reference sites across the NE Illinois IPS study area.

is more or less a permanent condition with little prospect of being abated quickly. Minimizing the amounts of salt that are applied over time is the most realistic best management practice for controlling this pollutant at present.

*Conductivity*

Conductivity measured as specific conductance can closely track chloride levels, but may also become elevated due to the presence of other dissolved ions. Conductivity threshold values (based on fish) in the NE Illinois IPS study area ranged from 739 μS/cm for the Exceptional threshold to 1038 μS/cm for the Good threshold (Table 8) which is close to values reported for Ohio (statewide; Ohio EPA 1999) and Southwest Ohio (MBI 2015).

*Total Dissolved Solids (TDS)*

TDS oftentimes tracks closely with chlorides and conductivity in urban watersheds, but it can be elevated by sources other than deicing chemicals. The database was sufficiently representative of all five narrative categories that an effect threshold could be derived for each (Table 8). There is a TDS criterion in the Illinois WQS which at 1500 mg/L is well above the very poor threshold of 744.5 mg/L. This WQS has also been outdated by newer data. It is part of a criterion that emanates from the 1970s era water quality criteria compendia (NAS/NAE 1973) that provided the basis for the initial state adoption of water quality criteria. The Exceptional threshold of 453.8 mg/L and Good threshold of 558 mg/L are somewhat higher than the corresponding values for similarly sized streams and rivers of 384 and 395 mg/L, respectively, in Southwest Ohio (MBI 2015).

*Sulfate*

Sulfates can be naturally occurring or the result of municipal or industrial discharges. Naturally occurring sulfate is oftentimes the byproduct of the decaying leaves that fall into streams and rivers, but can also be derived from the soil or from atmospheric deposition. Point sources include sewage treatment plants and industrial discharges such as tanneries, pulp mills, and textile mills. Runoff from the application of fertilizer to agricultural lands can also contribute sulfates to rivers and streams. The range of sulfate levels among the five narrative categories was low ranging from 58.3 mg/L for the Exceptional threshold to 93.8 mg/L for the Very Poor threshold, the latter of which is much lower than the equivalent Good value of 334 mg/L observed in the Western Allegheny Plateau (WAP) Level III ecoregion of Southeast Ohio where sulfates are elevated due the geology and acidic and non-acidic runoff from coal mining. This is an example of a parameter that, may need to be evaluated via alternate sources of data if it becomes a water quality issue in NE Illinois. However, all of these values are well below the 1809 mg/L criterion of the Illinois WQS.

***Demand Parameters***

Demand parameters include indicators of the presence and effects of organic enrichment that are primarily associated with carbon-based substances discharged by point sources and in runoff from organic rich soils. It is distinguished from nutrient enrichment (phosphorus and nitrate) although there is overlap between the sources and effects of these categories. For the

IPS, this parameter category includes dissolved oxygen (D.O.), 5-day biochemical oxygen demand ($BOD_5$), volatile suspended solids (VSS), ammonia-N ($NH_3$-N), and total Kjeldahl nitrogen (TKN), even though the latter two parameters are nitrogenous in origin and are frequently included with nutrients. Ammonia-N is a common wastewater constituent and TKN is the organic fraction of nitrogen and it can serve as a proxy for BOD (Miltner 2018). Suspended solids are also included in the Demand category and are represented by total suspended solids and turbidity. Among this category of parameters, only D.O. and ammonia-N have water quality criteria in the Illinois WQS. These will usually supersede the IPS derived thresholds for direct regulatory purposes (i.e., NPDES permitting) unless there is a compelling reason to do otherwise. Regardless, the IPS thresholds for total $NH_3$-N and D.O. can be useful for screening and planning purposes within the IPS framework.

*Dissolved Oxygen (D.O.)*

The minimum, maximum, and mean D.O. thresholds in Table 8 are based on daytime grab samples, thus they are best used as approximate indicators of D.O. depletion or excessive diel D.O. swings. Minimum and mean D.O. values along the gradient of the five narrative categories showed the expected relationship of declining values with declining narrative condition categories (Table 8). Maximum D.O. showed the opposite pattern with threshold values increasing with declining narrative condition a reflection of the likely effect of nutrient enrichment and other factors resulting in an increasing width of diel D.O. swings. These results are useful for at least two purposes; screening for sites and reaches with D.O. depletion issues and sites and reaches with excessive D.O. swings.

Ultimately, due to the complexities of the Illinois D.O. standard, continuous long term D.O. data will be needed to determine compliance. Short-term deployment of Datasondes during summer low flow periods can be effective for documenting the extent and severity of diel D.O. swings for assessing the effects of nutrient enrichment along with benthic and sestonic chlorophyll a (Ohio EPA 2015; Miltner 2018). This combination of data has been collected by certain watershed groups since 2017, but is currently insufficient to build the stressor threshold relationships that are needed for the IPS framework. Continuous data is needed from a sufficient number of Excellent and Good performing sites to yield valid stressor:response relationships.

*Ammonia-N ($NH_3$-N)*

The Illinois WQS for total $NH_3$-N t is applied based on the corresponding pH and temperature, as these determine the fraction of total $NH_3$-N that is present as the toxic unionized form of $NH_3$-N. The fraction of unionized $NH_3$-N increases with increasing temperature and especially increasing pH. Measured as total $NH_3$-N this parameter also showed a logical relationship across the five narrative condition categories (Table 8), but the correspondence to the fraction as unionized $NH_3$-N is weak with hardly any values resulting in an exceedance of the chronic water quality criterion. As such the IPS thresholds are used more as a screening indicator as opposed to an absolute determinant of instream exceedances of the WQS. WWTP compliance with water quality based effluent limits has been a key aspect of the general recovery of many rivers and streams that receive large volumes of municipal wastewater, when heavily polluted conditions of the pre-CWA era included $NH_3$-N as a leading cause of severe impairment. Capmourteres et al. (2018), in an analysis of a 1972-76 dataset that reflected pre-CWA water quality based controls, found $NH_3$-N to be a significant variable, among several, that explained degraded fish assemblages across NE Illinois. Much of that study area overlaps with the NE Illinois IPS study area. The IPS derived thresholds will likewise function as indicators of Susceptibility and Threat for attaining sites and reaches.

*5-Day Biochemical Oxygen Demand ($BOD_5$) and Volatile Suspended Solids (VSS)*

$BOD_5$ showed a logical relationship across the five narrative condition categories (Table 8) with reference values at or close to the minimum detection limit (MDL). While there is no WQS for $BOD_5$, NPDES effluent limits for WWTPs are based on meeting instream D.O. criteria. Since the larger point source loadings of $BOD_5$ are largely controlled, even the poor and very poor threshold values are well below historically polluted (pre CWA) conditions. Miltner (2018) included $BOD_5$ as a parameter related to nutrient enrichment in the Ohio large rivers assessment protocol. Still, since elevated levels can be a reflection of organic enrichment, which in urban areas can include domestic sewage from sewer overflows and leakage of sewage into stormwater, the detection of $BOD_5$ still has some relevance. Volatile suspended solids (VSS) can serve as a proxy for BOD as it represents the amount of volatile matter present in a sample. VSS also showed a logical relationship across the five narrative condition categories (Table 8), but with a lower FIT score than $BOD_5$, possibly due to a smaller and geographically restricted sample size.

*Total Kjeldahl Nitrogen (TKN)*

Total organic nitrogen as measured by Total Kjeldahl Nitrogen (TKN), an indicator of the living or recently dead fraction of sestonic algae, is an informative indicator of both organic and nutrient enrichment. While TKN is not a direct effect parameter, it is indicative of the effects of organic enrichment by nitrogenous biomass. Major sources of organic nitrogen in urban stormwater runoff include lawn and garden fertilizers, pet waste, leaking septic tanks, landfills, effluent from sewage treatment plants, and vehicle exhaust (U.S. EPA 2020). Nitrogen from aerial and terrestrial sources accumulates on urban roads and parking lots until runoff from a precipitation event carries the pollutants into stormwater drains and directly to local waterbodies. Among different land uses, the highest concentrations of TKN originate from impervious surfaces (e.g., freeways, parking lots, and high density residential). There is no WQS, but IEPA regards it as an important parameter to monitor instream. Miltner (2018) regards TKN as a proxy for BOD, given its value and an indicator of overall enrichment particularly by algal biomass. While it has at least a directional relationship across the five narrative condition categories, the spread between Excellent and Very Poor values is not particularly great (Table 8). However, the regional reference values are likely a better reflection of what constitutes low TKN values. This is a parameter where exceedances of a narrative threshold do not necessarily correspond to a biological impairment, but in combination with other parameters can be indicative of excessive nutrient or organic enrichment particularly from urban sources (U.S. EPA 2020).

*Total Suspended Solids and Turbidity*

Total suspended solids (TSS) and the closely allied turbidity (NTU) reflect the concentration of suspended materials in the water column. Such materials can be a mix of inorganic and organic materials, and the composition of each can reflect differing sources and effects. Various types of suspended matter in the water column of streams have been associated with impairment to aquatic life, and the relationship can be complex. Newcombe and Macdonald (1991), in a review of available research papers, concluded that the concentration of suspended materials alone is a weak indicator of adverse effects and that the duration of exposure to elevated concentrations needs to be considered. TSS had a directional relationship across the five narrative condition categories although the spread between Excellent and Very Poor values is not particularly great (Table 8). Anchoring the high end expectations in the regional reference median TSS value would improve that spread, but the interquartile range showed a wide variability encompassing the Excellent and Good thresholds. The thresholds for turbidity exhibited a similar pattern. In combination with other parameters, these can be useful secondary indicators, but as standalone parameters, they are not especially useful. Measures of the harmful effect of the deposition of suspended materials as reflected by the QHEI substrate metric scores are more strongly associated with biological condition and likely reflects the need, as recommended by Newcombe and Macdonald (1991), to consider the duration of elevated TSS and turbidity. TSS is frequently used as a primary stormwater indicator and on a standalone basis in some cases. TSS needs to have the "tempering" of habitat measures that include metrics that are responsive to sedimentation. Given the rather weak performance of TSS as a useful indicator in the NE Illinois IPS, a more comprehensive suite of indicators are needed to reliably manage and assess the impacts of stormwater.

***Metals and Toxics***

This category includes the more commonly encountered heavy metals, and other toxic substances, included in the rotating watershed monitoring conducted by the watershed groups and IEPA in the NE Illinois IPS study area. Elevated concentrations of metals and other toxicants in the water column are relatively uncommon compared to conditions that existed prior to CWA mandated controls on point source discharges. Nevertheless, these parameters can still be detected locally and particularly via concentrations in sediment. Most of these parameters have Illinois WQS that would supersede the IPS thresholds for regulatory purposes such as NPDES permitting. The IPS thresholds are intended to be used in a lines-of-evidence application for diagnosing aquatic life impairments and for contributing to the Susceptibility and Threat scores. The low frequency of >MDL concentrations for these parameters hinders the derivation of effect thresholds because levels that drive negative biological responses are very infrequent. For example, total copper in the NE Illinois IPS study area (Figure 21, right) has only three (3) values of copper >10 µg/L and very few values between 6-10 µg/L compared to a similar analysis in Southwest Ohio that had more pre-CWA historical data with elevated total copper values. While the lack of elevated copper values in the NE Illinois IPS study area is a positive observation for water quality, the lack of a sufficient range of values across the spectrum of the biological narrative condition categories can deter the derivation of meaningful thresholds. The

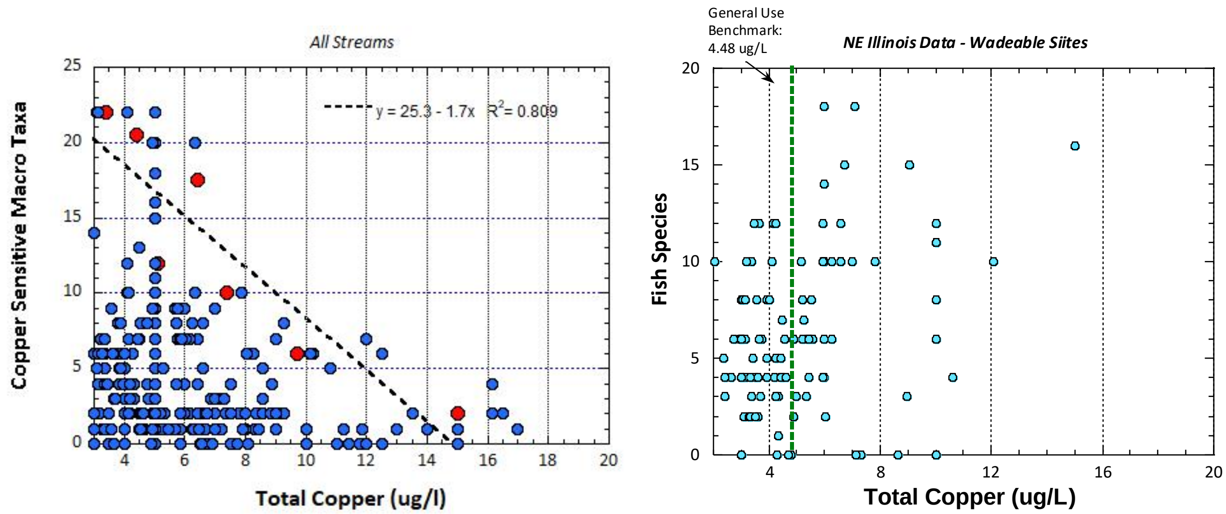



***Figure 21****. Plot of total copper vs. sensitive fish species in NE Illinois rivers and streams (right) compared to a similar analysis done for Southwest Ohio (left; MBI 2015) where a wider range of copper concentrations occurred.*

inclusion of historical copper data with paired with fish and/or macroinvertebrate data would likely improve the derivation of more realistic thresholds. The majority of the metals and cyanide had FIT scores ranking in the lower half of the 31 parameters in Table 8. An exception was zinc which had the fifth highest FIT score out of 31 water column parameters (Table 8). The Illinois criteria normalized to a hardness of 300 mg/L are included in Table 8 for reference and should be used in lieu of the IPS thresholds for determinations of causes impairment.

## Nutrient Parameters

Nutrients, and the development of nutrient water quality criteria for streams and rivers are currently among the most emphasized of near term WQS and regulatory developments. For this and other reasons, the derivation of meaningful effect thresholds is a high profile issue for NE Illinois watershed stakeholders. While there are demonstrated associations between nutrient parameter concentrations and aquatic life (Ohio EPA 1999, Miltner 2010), the relationship is complex because the mode of effect is largely indirect (Miltner 2018). The State of Illinois Nutrient Science Advisory Committee (NSAC) developed the Illinois Nutrient Loss Reduction Strategy (NLRS; NSAC 2018) to deal with the enrichment of Illinois surface waters by primary nutrients (N and P). As part of the NLRS, IEPA developed a process termed the Nutrient Assessment Reduction Plan (NARP) which is to be developed for major WWTPs by 2023 and in some cases by 2024. All of the major WWTPs that are members of the DRWW, NBWW, DRSCW, and LDRWC have initiated planning projects for meeting the NARP requirements. Depending on the findings of the NARP process additional controls on discharges of N and P could be forthcoming. Nutrient parameters are those associated with the potential effects of eutrophication due to increased algal activity and can arise from changes in species/taxa composition based on changing trophic dynamics or, when more severe, from changes due to shifts in parameters such as D.O. and pH.

Efforts are currently underway in several states to develop what are termed combined nutrient criteria that utilize a mix of direct and effect based indicators to evaluate the consequences, risks, and severity of nutrient enrichment. Some of the most important measures of the effect of nutrient enrichment are the subject of ongoing research to develop more meaningful nutrient effect thresholds across NE Illinois, particularly for D.O. (MBI 2023). Until that research is completed, an initial reliance on IPS effect thresholds for key nutrient parameters and an in-development Stream Nutrient Assessment Procedure (SNAP; Ohio EPA 2015) will need to suffice, the latter for watershed assessment purposes.

The approach to nutrient management by point sources, specifically municipal WWTPs, is affected by recent initiatives of IEPA, most notably Nutrient Assessment Reduction Plans (NARP) and Nutrient Implementation Plans (NIP), the two of which are synonymous. These plans are to be developed by WWTPs upstream of waters with any impairment linked to phosphorous (i.e., listed in Section 303 (d) with phosphorous as a cause) or at "risk of eutrophication" (with thresholds for pH, D.O., and sestonic algae). These plans can be developed individually or jointly as part of a watershed group. All of the major permitted WWTPs in the NE Illinois IPS project area are subject to Special Conditions related to the discharge of nutrients and the NARPs/NIPs, but not all have final language.

***Nutrient Assessment Reduction Plan (NARP)***

The first special condition relates primarily to feasibility studies that identify the method and costs of reducing phosphorus levels in a discharge to a potential future effluent standard of 0.5 and 0.1 mg/L, on a monthly, seasonal, and annual average basis. Most of the watershed workgroups are engaged in modeling to simulate P loadings with the D.O. regime in order to evaluate the feasibility, and cost of the aforementioned effluent standards. All of the workgroups have access to the IPS derived threshold for total P and some are using it as an option for determining TP effluent limitations within the bounds of the disclaimer at the beginning of this document. The SNAP methodology (as modified from Ohio EPA 2015) has been applied consistently in only the DRWW and NBWW watersheds and represents yet another option for addressing NARP.

***Nutrient Implementation Plans (NIP)/Nutrient Implementation Plan (NARP)***

The second special condition deals with the submittal of a Nutrient Implementation Plan (NIP) or its equivalent Nutrient Implementation Plan (NARP), for IEPA approval with the NPDES renewal application. The NIP must identify phosphorus input reductions by point source discharges, non-point source discharges, and other measures necessary to remove D.O. and offensive condition impairments in a watershed. A WWTP may work cooperatively with their Watershed Workgroup to prepare a single NIP that is common among all NPDES permittees. In the DRSCW and LDRWC areas this provision is met by the Basin Bioassessment Plan.

***Deriving N and P Thresholds***

The identification of nutrient-sensitive fish species and macroinvertebrate taxa improved the sensitivity of the response to N and P concentrations (Appendix C). However, neither the nutrient sensitive fish species nor macroinvertebrate taxa are a precise "fingerprint" of nutrient impacts, but more of a "smudged" thumbprint. This is because most species/taxa are sensitive to multiple stressors some of which correlate with nutrient enrichment. The FIT scores derived later relate to the precision of the stressor fingerprints. Note that the plots of the mIBI or fish IBI with the stressor-sensitive species/taxa are nearly always more distinct (i.e., strong wedge shaped plots with sharp threshold responses) than the plots of the stressor itself with the sensitive species/taxa (Appendix C). This is partly due to the fact that some species that are sensitive to total P as measured by the weighted means are also sensitive to other related parameters such as TKN and habitat. Thus, and as the mIBI or fIBI declines, some of the species and taxa contributing to the reduced index scores could be responding to other stressors. However, the effects are frequently allied such that species that are sensitive to phosphorus are not likely to also be tolerant of low D.O.

It is expected that the statistical power of these relationships will improve as more data is accrued via routine monitoring and assessment and as "layers" of stressors are reduced as BMPs and other management interventions are implemented. The causal analyses in Chapter 4 provide a more complete comparison of stressor categories and their influence on aquatic life (i.e., nutrients vs. habitat. vs. dissolved solids vs. land use/altered flow measures vs toxicants in water and sediment). Even with the limitations for precisely allocating effects between parameters and parameter groups, total P especially is logically associated with biological impairment with a strong directional relationship across the five narrative condition categories, with a relatively large spread between the regional reference median, the Excellent and Very Poor values, and the highest FIT score among the water column parameters (Table 8). For example, TP concentrations in rivers and streams with fIBI scores >50 are distinct from the other IBI narrative ranges. A simple Analysis of Variance (ANOVA) confirms this (F Statistic = 3.04; $P<0.019$) with the TP values in the Excellent range contributing the most to this result. The relationship with N as nitrate-N also had a logical and directional relationship across the five narrative condition categories, but the Excellent threshold value was nearly one order of magnitude higher than the regional reference median (Table 8).

***Nutrient Effects Assessment***

The impact of nutrients on aquatic life has been well documented (e.g., Allan 2004), but the derivation of criteria and their form and application are only just now emerging. Because of the widely varying efforts to develop nutrient criteria by the States, conflicting U.S. EPA oversight, and the potential cost of additional nutrient controls it is a controversial issue (Evans-White et al. 2014). Unlike toxicants, the influence of nutrients on aquatic life is indirect and primarily via their influence on algal photosynthesis and respiration and the resulting increased magnitude of diel D.O. swings and by the biochemical oxygen demand exerted by algal decomposition. Nutrients can also affect food sources for macroinvertebrates and fish, and the response of aquatic life to elevated nutrients is co-influenced by habitat (e.g., substrate composition), stream flow (e.g., scouring and dilution), temperature, and exposure of the water column to sunlight. Illinois is the leading state in terms of nitrogen (16.8%) and phosphorus (12.9%) loadings exported towards the Gulf of Mexico via the Illinois and Upper Mississippi Rivers where an anoxic zone has developed (U.S. EPA 2008). In Illinois, as in neighboring Midwestern states that drain to the Mississippi River, efforts are underway to modernize nutrient water quality criteria. However, nutrient export is not the only concern – local impacts are also important, and the focus of the IPS is on local scale effects in NE Illinois watersheds.

A new methodology to assess the effect of nutrient enrichment was introduced in the 2017 Year 1 assessment for the DRWW (MBI 2019). Modeled after the Stream Nutrient Assessment Procedure (SNAP) developed by Ohio EPA (2015), it includes consideration of the width of the diel variation in continuously measured D.O., the biomass of chlorophyll a in benthic algae in addition to the concentration of total phosphorus and dissolved inorganic nitrogen (nitrates + nitrites). Other nutrient related parameters such as volatile suspend solids (VSS), turbidity, and total Kjeldahl nitrogen (TKN) were included when they were collected at one of the Datasonde and benthic chlorophyll-a locations. Datasondes were deployed for consecutive 3-4 day periods during times of low stream flow and elevated summer ambient temperatures (YSI 2012, 2017). New to this analysis in 2018 are the number of phosphorus sensitive species derived from the NE Illinois IPS stressor analyses and a Nutrient Ranking Index that was also developed with IPS outputs (Appendix D). Together these results were used to determine five degrees of nutrient enrichment (none, low, moderate, high, and severe).

### *Northeast Illinois IPS Nutrient Ranking Index*

To further support the assessment of nutrients, a Nutrient Ranking Index (NRI) was developed by summing the ranking of each of the individual primary nutrient or nutrient-related parameters with each weighted based on the FIT coefficient (Table 11). The equation follows:

Nutrient Rank Index = (TPR*1) + (Min. DOR*1) + (TKNR*0.8) + (BODR*0.8) + (NITRR*0.8) + (Max. DOR*0.6)

Where; TPR = Total Phosphorus Rank
Min. DOR = Minimum Dissolved Oxygen Rank
TKNR = Total Kjeldahl Nitrogen Rank
BODR = Biochemical Oxygen Demand (5-day) Rank
NITRR = Nitrate Rank
Max. DOR = Maximum Dissolved Oxygen Rank

***Table 11. Fit weighting scores based on FIT coefficients. FIT measure discussed in Chapter 4.***

| |
|---|
| **FIT (< 0.10) X 1;** |
| **FIT (> 0.10 – <0.3) X 0.8** |
| **FIT (> 0.30 – < 1.0) X 0.6** |
| **FIT (> 1.00 – < 3.0) X 0.5** |
| **FIT (> 3.00 – < 10.0) X 0.2** |
| **FIT (> 10 0) X 0.1** |

Figure 22 illustrates the correlation between the NRI and the fIBI (top, left), mIBI (top, right), the number of Illinois intolerant fish species (bottom, left) and the number of Illinois intolerant macroinvertebrate taxa (bottom, right). In these graphs, the points were coded to the strongest stressor rank for all categories of stressors (excluding land use parameters) and where the most limiting stressor rank was greater than four (i.e., General Use benchmark). Boxes in the upper right corner reflect NRI ranges where biological performance is clearly limited. In these plots, fish appear slightly more limited than macroinvertebrates. We expect the relationship between the NRI and biological response variables to improve as other indicators such as continuous dissolved oxygen-based maximum daily D.O. swings and algal indicators (benthic chlorophyll) are better

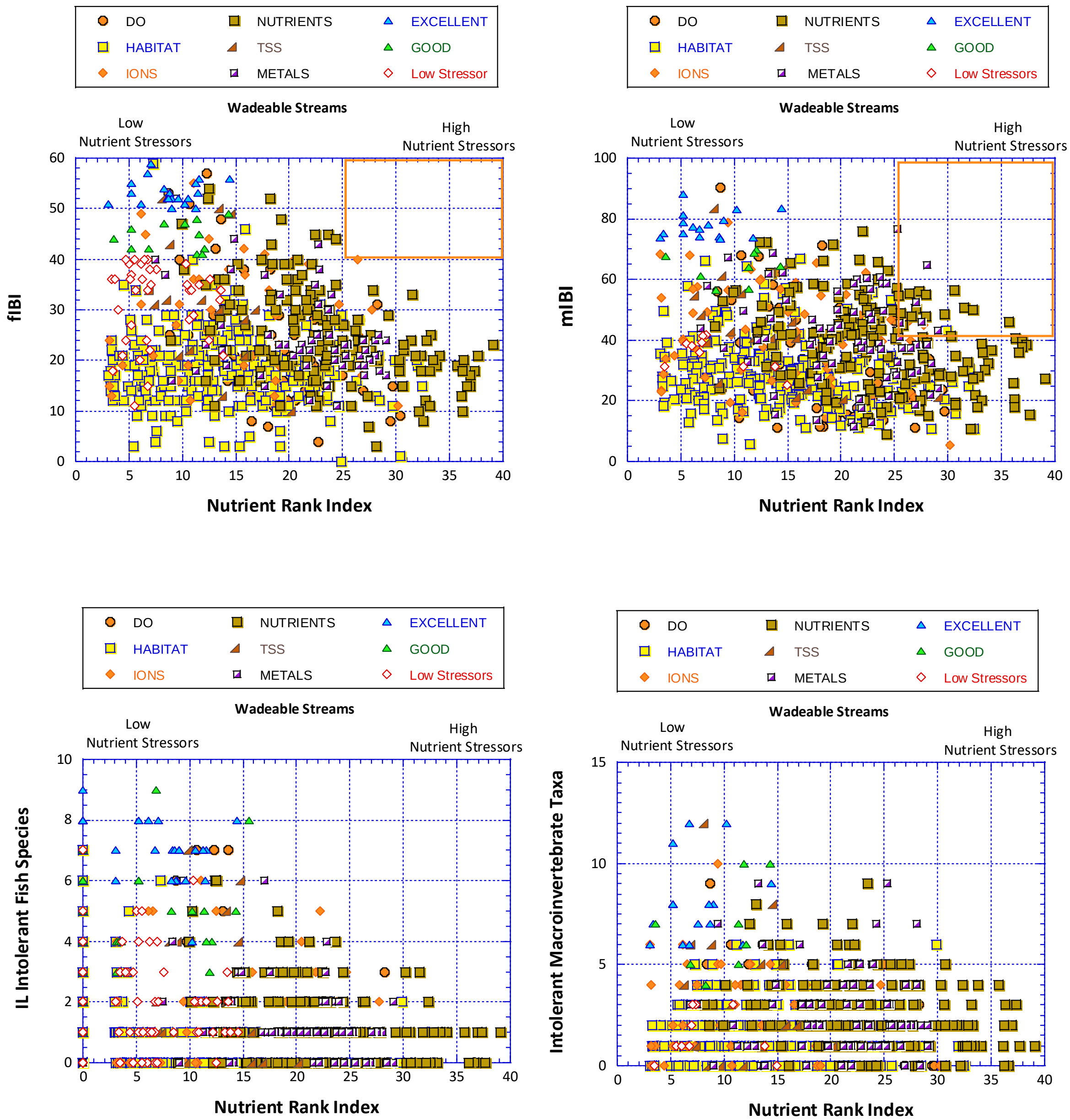



***Figure 22.*** *Correlation between the Nutrient Ranking Index and the fIBI (top, left), mIBI (top, right), the number of Illinois intolerant fish species (bottom, left) and the number of Illinois intolerant macroinvertebrate taxa (bottom, right). In these graphs, points are coded (see legend) by the strongest stressor rank for all categories of stressors (excluding land use) and where the most limiting stressor rank was greater than a score of four (i.e., General Use benchmark).*

developed. Even so, there is a strong enough relationship to make the NRI a useful indicator of eutrophication in a study area. NRI values of >25 are always associated with impaired fish assemblages and often associated with impaired macroinvertebrates (Figure 22). Where a biological assemblage is of Excellent quality, NRI values are nearly always <15. The Power BI dashboard for nutrients will provide this data for all sites where it is available and will also provide individual parameter rankings for nutrients and other parameter categories (e.g., TP, TKN, min D.O.) as well. Such data can be matched to local continuous D.O. and benthic and sestonic chlorophyll a data where it exists and SNAP results where that analysis has been conducted. For example, sites with high NRI values and elevated D.O. swings from continuous data can be examined along with biological assemblage responses to determine if the patterns of response are similar. The IPS Dashboard will also have NRI values, among other data, summarized at the reach and HUC12 scales to determine the prevalence of nutrient signatures nearby and across a watershed. The goal for developing the NRI is to have a screening value that can then be matched to site specific data to conduct a stressor identification analysis.

## Sediment Chemistry Variables

Toxic substances in sediments can be associated with aquatic life impairments in areas with moderate to high development (e.g., streets, roads, parking lots) or as a legacy effect of past industrial activities (manufacturing, mining, etc.). The IPS focuses on heavy metals and polycyclic aromatic hydrocarbons (PAHs) in sediment because of their consistent presence in the urbanized watersheds of the IPS study area. Sediment chemical screening guidelines have been developed with the most common being the consensus-based (i.e., the average of multiple thresholds compiled from different studies; Persaud et al. 1993; CCME 1999; McDonald et al. 2000; NOAA 2008). The MacDonald et al. (2000) Threshold Effect Concentration (TEC) and Probable Effect Concentration (PEC) cover the most compounds while the Lowest Effect Level (LEL), Threshold Effect Level (TEL), Probable Effect Level (PEL), and Severe Effect Level (SEL) guidelines (Persaud et al. 1993; CCEM 1999) cover additional compounds. At concentrations below the TEC, TEL, or the LEL, effects on the most sensitive macroinvertebrate taxa are expected. For concentrations exceeding the PEC, PEL, and SEL thresholds, adverse effects are expected for all taxa. IPS thresholds were derived in this study based on the response of contaminant-specific sensitive macroinvertebrate taxa. A shortcoming with the IPS sediment chemistry database is the comparative lack of data at sites with Excellent mIBI scores resulting in thresholds being developed mostly for sites with Good mIBI scores. While there was insufficient data to generate regional reference values at this time, data is now being collected by DRSCW and LDRWC at reference sites. Once sufficient data is available the IPS derivation of PAH thresholds can be revisited. Threshold analyses for fish was not performed as there were essentially no sites with attaining fIBI values, but much of the literature about the effects of sediment contaminants have focused on macroinvertebrates as the most sensitive assemblage simply because of their close contact with bottom sediments. Another caution about applying sediment chemistry data is about what it represents in a stream or river. Rarely, if ever, are sites “blanketed” with sediments containing toxic compounds. Rather, the sediment is present as a “ribbon” along the shoreline and the analysis is of the clay and silt fraction of a sediment sample. As such it is a reflection about levels of these chemicals as they pass through the stream or riverine system, not necessarily as deposits extensive enough to require extraction to remediate adverse impacts.

### *Polycyclic Aromatic Hydrocarbon (PAH) Compounds*

All of the detected PAH compounds are byproducts of the incomplete combustion of organic materials and several are known carcinogens. These compounds are commonly detected in urbanized watersheds with large areas of asphalt pavement and heavy automobile traffic and presumably enter streams via runoff from paved surfaces. Some PAH compounds result from the incomplete combustion of hydrocarbons and are a common component of stormwater runoff in urban areas – they are not a direct byproduct of any manufacturing process. For most of the common PAH compounds, the IPS thresholds (Table 9) were in between the TEC/TEL/LEL levels and PEC/PEL/SEL guidelines, but closer to the TEC/TEL/LEL guidelines. Most IPS thresholds below TEC/TEL/LEL levels were nonetheless very close to those thresholds. For the most part, the IPS derived thresholds are in agreement with effect levels of the consensus screening benchmarks listed in Table 9. In watershed assessments, both the IPS and consensus thresholds should be used especially since the availability of IPS thresholds is not yet complete for all of the PAH compounds.

***Sediment Metals***

Elevated levels of heavy metals in sediment are commonly associated with runoff from roads and highways and industrial and municipal sources. As with PAHs, sediment metal detections are the most widespread in urban areas being the result of urban and industrial runoff and runoff from road surfaces. As with PAHs, most of the IPS sediment metals thresholds (Table 9) were close to or even slightly above the TEC or LEL consensus thresholds. Excellent thresholds could be derived for only five of the most commonly monitored metals, and detections are not expected at Excellent performing sites. None had particularly strong FIT scores with most ranking in the lower one-third of all stressors in Tables 8-10. This is likely a result of the absence of data at Excellent performing sites and data indicative of the acutely toxic conditions that were more widespread before water quality based CWA pollution controls were imposed. IEPA (Short 1998) developed elevated and highly elevated sediment values for 12 metals based on a statewide distribution of sediment results that are also used to screen sediment results in watershed assessments (Table 9). These are not effect based levels.

# Chapter 4 Causal Associations

## INTRODUCTION

For rivers and streams identified as having impaired aquatic life uses, a major function of the IPS is to enhance not only the identification of causes and sources of impairment, but demonstrate their comparative severity in terms of exceeding the Fair, Poor, or Very Poor SSD thresholds. Given that multiple stressors that can affect aquatic life at a single site or in a single stream reach, particularly in urban settings, there is a practical need to narrow causes down to the most critical ones. FIT analyses and Random Forest regression and classification tree analyses were each used to assess the strength of the response of the aquatic biota to a stressor and enable ranking the most important causes as a result. There was a need to reduce the number of stressors prior to using the multivariate statistical methods. This reduction was first achieved by correlation and simple regression and classification trees by stressor category.

### Causal Analysis and Variable Reduction Approaches

This chapter focuses on the weight-of-evidence for casual associations between the key response variables and individual and categorical stressors. A key aspect of using the SSD effect thresholds is having the ability to distinguish those that are likely responsible for observed biological impairments from those that are merely correlated. For example, a large number of land use variables were compiled and analyzed, but most are highly correlated with one another. Land use variables are more properly considered as “source” measures with the actual stressor “agents” (causes) as their “product” including altered flow regimes, habitat alterations, and increased urban pollutants, all being delivered more quickly and in larger quantities to affect habitat and water column and bottom sediment quality, each of which can elicit adverse and sometimes distinctive biological responses. Three types of analyses were used to identify the most important causal variables for the IPS:

1) Correlation analyses and regression and classification tree analyses by stressor category using the entire dataset to reduce variables and remove highly correlated variables;
2) Derivation of a goodness-of-fit (FIT) statistic to measure the strength of actual vs. predicted associations between parameters and sensitive species/taxa richness; and,
3) Random forest regression and classification analyses on the subset of the data produced by the correlation and regression tree analyses to better discriminate the level of importance among the reduced set of variables.

Parameter importance was also tempered by the derivation of the SSD effect thresholds, controlled studies certain parameters, information in the ecological literature about the importance of the effects exerted by a particular variable, and inferences based on the iterative watershed monitoring supported by the watershed groups. A weakness of the multivariate analyses is the potential rejection of one or more variables that actually have meaningful effects. Including the results of the above can better assure that important effects are included.

The SSD thresholds derived in Chapter 3 were used as factors in the Restorability scores with their contribution weighted by the magnitude of stressor effects and the number of stressors with elevated stressor levels. Analyses that distinguish among variables that are the strongest correlates with biological condition and response can be used to augment the weighting of stressors that are used in the Restorability and Susceptibility and Threat ratings. The selection of the "strongest" correlates, however, does not eliminate the usefulness of multivariate analysis to better explain causality, particularly at the site-specific, reach, and watershed scales. With habitat, for example, the QHEI score is likely to be among the strongest causal variables in the regression and classification analyses. That is partly due to the fact that it integrates multiple attributes of habitat (e.g., substrate quality, cover, channel condition, etc.). It may be that individual habitat stressors provide more insight into what specific factors are actually the strongest causal variables (e.g., substrate quality) at a given site, than the QHEI score which over a range of sites may prove to be the most statistically significant variable ***on average***. In addition, while broad scale multivariate analyses provide insights into average or prevalent conditions, it can obscure the site-or reach-specific importance of an excluded variable.

## Discriminating Among Categories of Impact: Correlation Analyses and Classification and Regression Trees

In Chapter 3 SSD effect thresholds were derived for individual parameters regardless of the penchant for multicollinearity[11] across variables or categories of variables. Conceptual models, experimentation, behavioral ecology, and primary ecological studies are all important for understanding potential pathways and modes of stressor effects. With large enough datasets that have a wide enough "span" across environmental quality gradients, statistical analyses can be useful to identify limiting factors. Given the large number of environmental and stressor variables amassed in the NE Illinois IPS database, a three-step process was used to identify the most influential stressor variables and for identifying the mechanisms causing aquatic life impairments. These analyses should provide the variables that will be the most useful in selecting effective restoration and protection and setting goals for stressor reduction. For each category of stressors a correlation matrix was derived to reveal which variables are highly correlated defined here as an ~r value >0.7. Among the highly correlated variables the ones that were readily measured and that match conceptual models, and other evidence of causal impacts, were retained. Classification and regression trees were then used to identify the strongest among these variables for the derivation of FIT scores, random forest regression, and classification analysis. Variables with stronger correlations were given more weighting in the IPS Restorability factor.

### *Physical Habitat*

Physical habitat has been shown to be one of the most influential variables determining the condition of both fish and macroinvertebrate assemblages in Midwest streams and rivers. The QHEI is used widely for stream habitat assessments and it consists of an overall score which

[11] Multicollinearity is a statistical concept where several independent variables in a model are correlated. Two variables are considered perfectly collinear if their correlation coefficient is +/- 1.0. Multicollinearity among independent variables will result in less reliable statistical inferences.

results from a series of metric and attribute values that are summed to yield a QHEI score with a range of 0-100. Some QHEI metrics can be highly correlated with the overall index score, but each measures a distinct physical attribute (e.g., substrate type, channel form, cover, etc.) that provides insight into which attributes are limiting to aquatic life. Figure 23 presents the results of a correlation matrix of Pearson correlation coefficients and as expected many of the metric scores are highly correlated with the overall QHEI (r >0.7) with exceptions being the riparian metric (r= 0.39) and gradient score (r=0.23). The importance of the riparian metric is thought to be important to the cumulative condition of watersheds (May and Horner 2000), although the site-specific correlations are weaker than other habitat metrics (Figure 23). The gradient score

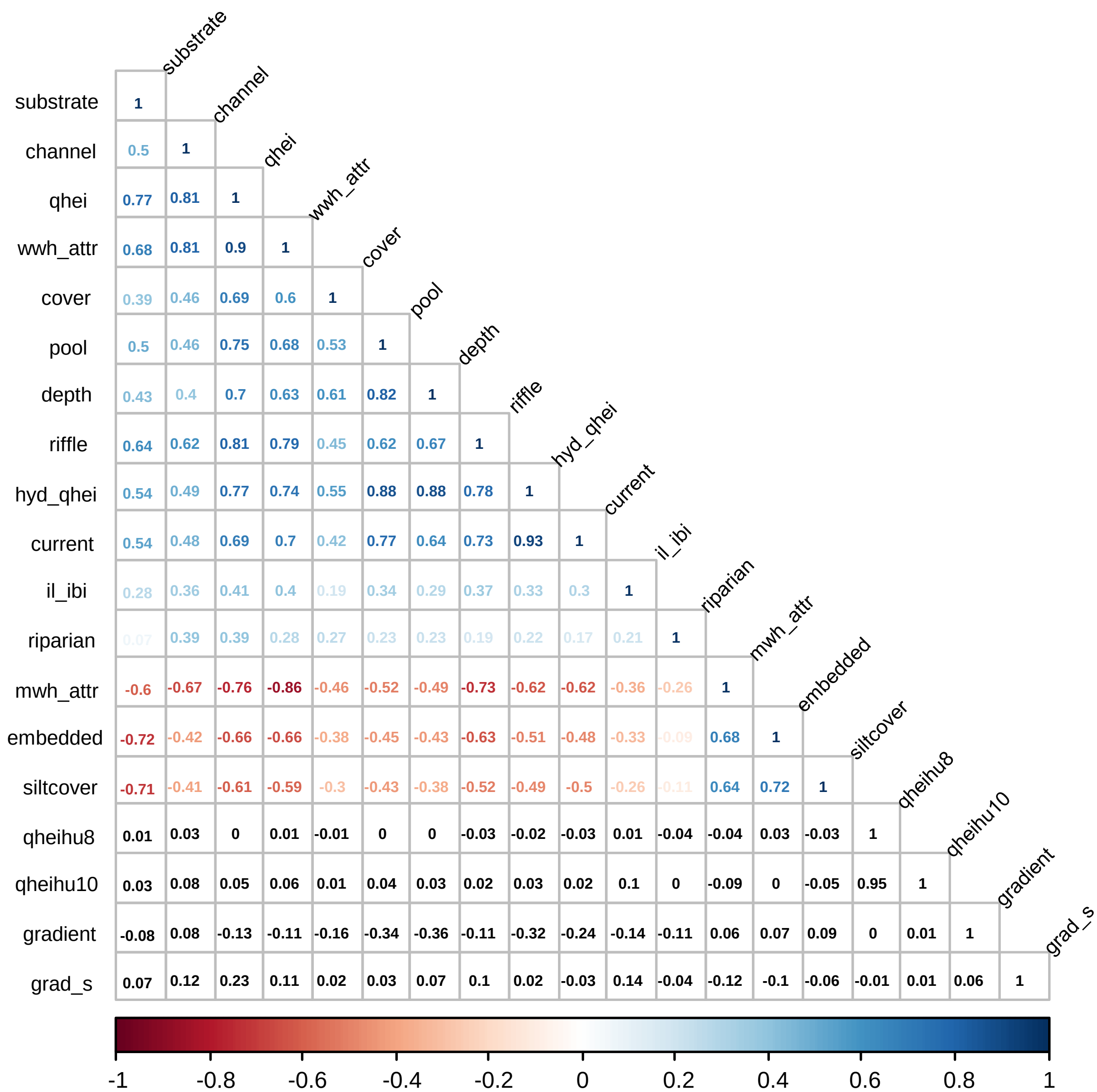



***Figure 23****. Matrix of Pearson correlation coefficients (r values) for physical habitat parameters (QHEI, QHEI metric and attributes, Hydro-QHEI, etc.) in the NE Illinois IPS study area. Highly correlated variables are clustered together. Positive correlations are blue and negative correlations red with color intensity greater with higher correlation coefficients.*

likely did not have a wide enough variation across the IPS study area to elicit a statistically meaningful result which is a reflection of a natural characteristic as opposed to an impact.

These results were then used in regression and classification tree analyses to explore which variables were most important in explaining variation in the fIBI and attainment of Good and Excellent fIBI thresholds. The nodes on each regression tree represent the mean fIBI and the proportion of the sites represented by that mode. The first split in the regression tree largely separated sites with higher fIBI scores from those with lower scores and split between sites with less than or greater than a QHEI score of 74 which is very close to the Good SSD threshold of 75.9 (Table 10). For sites with higher fIBI scores the next split is a QHEI >89 which groups most of the sites with fIBI scores $\geq$42. The rest of the nodes split on stream gradient or the QHEI gradient score (Figure 24).

Changing the response variable from attainment of the General Use for aquatic life to attainment of the Excellent narrative range of the fish IBI produced nearly identical classification trees (Figure 25). The first split was on the number of modified (poor) habitat attributes (score of 0.025) which essentially represents a natural channel with few or no modified habitat attributes. When the modified attributes were low then a high channel score separated attaining from impaired sites (Figure 25, top) or Excellent from lower quality sites. Based on the regression and classification tree analyses, QHEI score, modified (poor) habitat attributes, the channel metric, and gradient metric are, on average, the best overall explanatory habitat attributes for fIBI scores and whether or not the scores attain the General Use (Good) or Excellent fIBI thresholds. ***[Variables to be included in overall regression and classification tree:** QHEI, Stream Gradient, Modified Attributes, Channel Score**]**.*

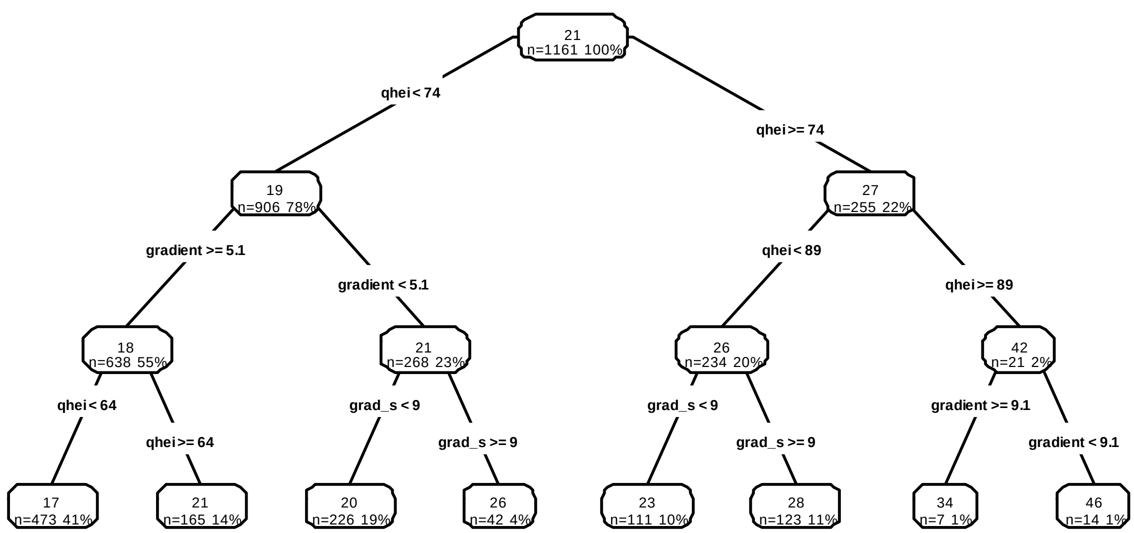



***Figure 24.** Results of a regression tree analysis with the Illinois Fish IBI as the response variable and physical habitat parameters (QHEI, QHEI metric and attributes, Hydro-QHEI, etc.) as explanatory variables. Because a goal was the reduction of parameters, the tree was limited to three levels.*

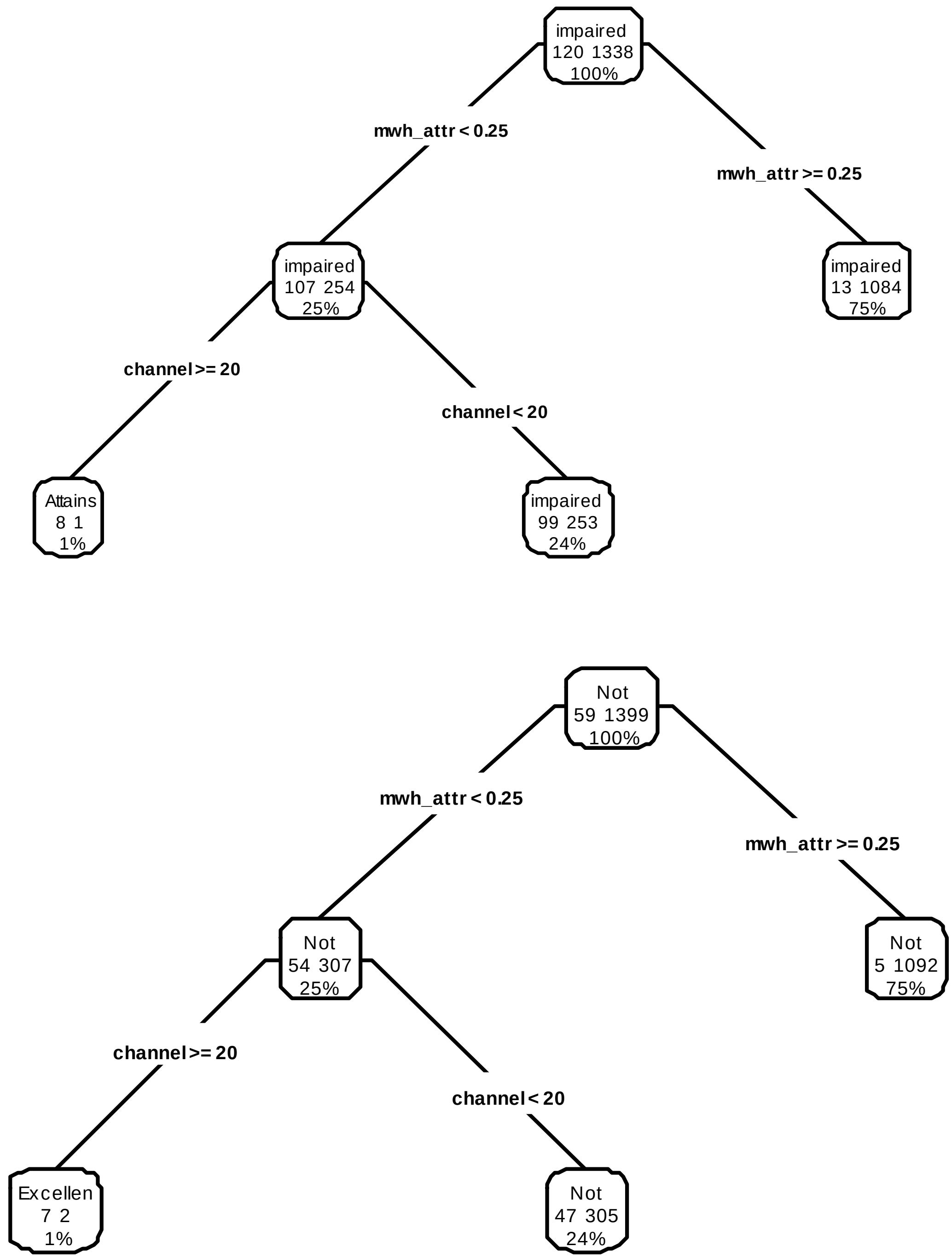



***Figure 25**. Results of two classification tree analysis with the attainment of the Illinois Fish IBI as the response variable (top) or attainment of the excellent narrative threshold (bottom) with physical habitat parameters (QHEI, QHEI metric and attributes, Hydro-QHEI, etc.) as the stressor values. Because a goal was the reduction of parameters, tree was limited to three levels***.

***Ionic Strength Parameters***

Ionic strength parameters were generally correlated with one another (Figure 26), but not as highly correlated as the QHEI metrics. Regression and classification tree analyses were conducted with all ionic strength variables (Figures 27 and 28) using the fIBI as the response variable because the most restrictive SSD thresholds were based on the fish assemblage response. The regression tree analysis used the fIBI as the response variable with two classification tree analyses using attainment of the General Use (Good) fIBI threshold score as one endpoint and the attainment of the Excellent fIBI threshold as another endpoint.

The first major split on the regression tree was on chloride levels less than or greater than or equal to 50 mg/L (Figure 27). This result matches the general concern expressed in the literature as a key variable in the salinization of freshwater (Miltner 2021). At the second and third levels of the regression tree, chloride (elevated concentrations) and conductivity were responsible for splits between the branches of that tree. Figure 28 illustrates a classification tree based on which ionic strength variables best explain attainment of the General Use fIBI (top) or the Excellent fish IBI narrative (bottom). Sites with chloride concentrations less than 38 provide the first split between sites that attain or do not attain the fIBI threshold (Figure 28,

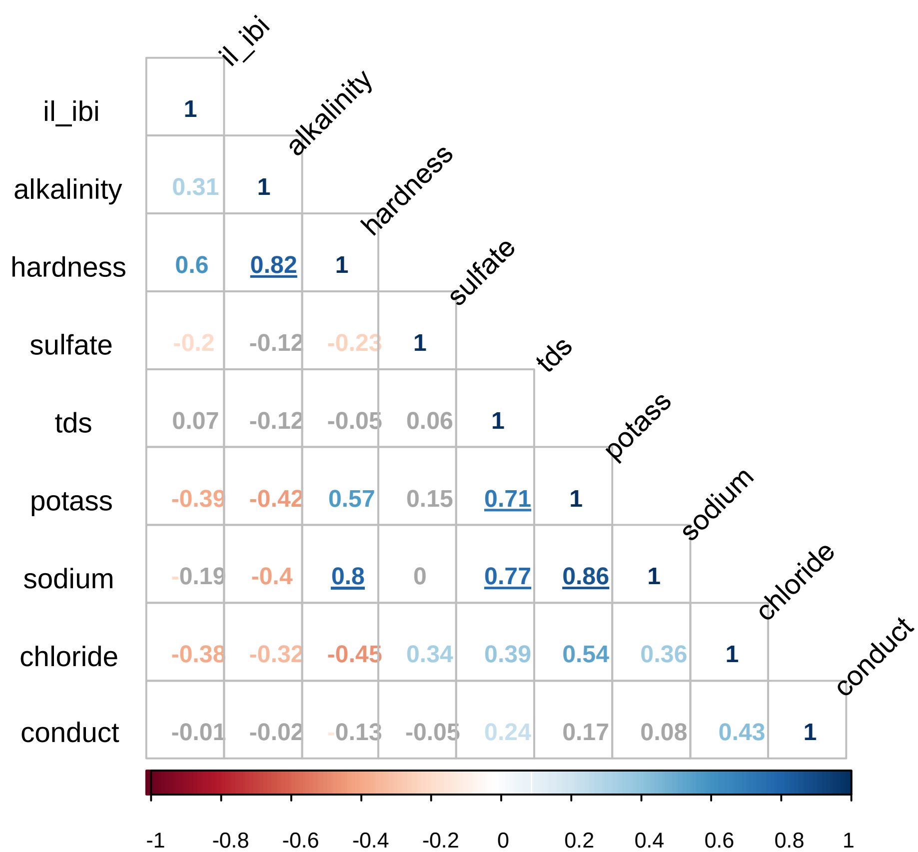



***Figure 26**. Matrix of Pearson correlation coefficients (r values) for ionic strength variables in the NE Illinois IPS study area. Highly correlated variables are clustered together. Positive correlations are blue and negative correlations red with color intensity greater with higher correlation coefficients; Coefficients > 0.7 are underlined.*

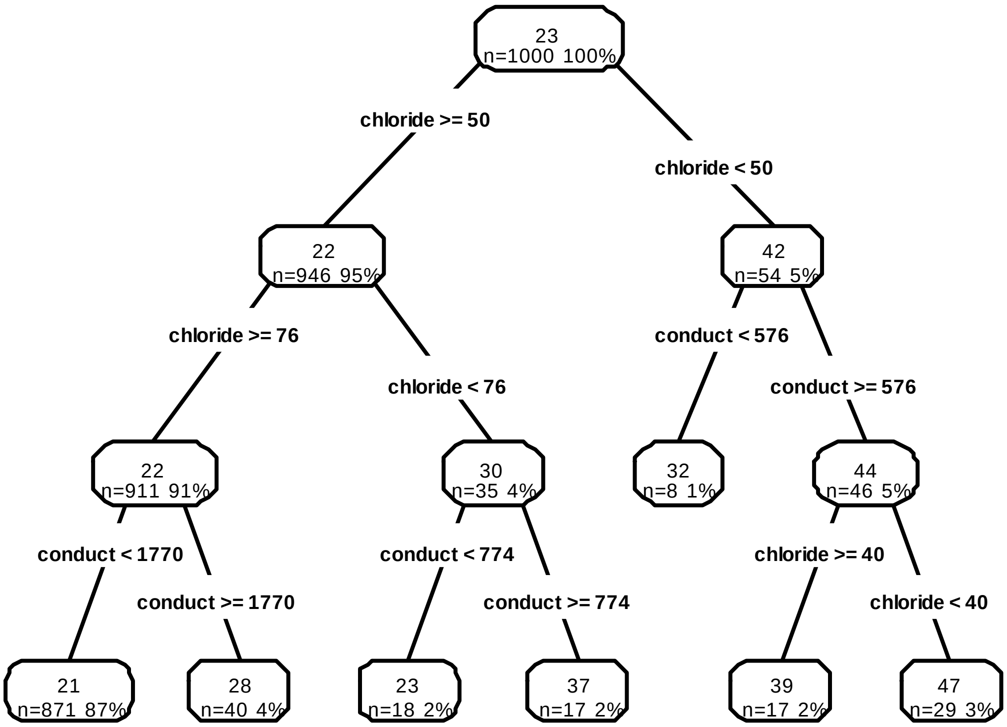



***Figure 27****. Results of a regression tree analysis with the Illinois fIBI as the response variable and ionic strength parameters (chloride, sodium, sulfate, potassium, total dissolved solid and conductivity) as explanatory variables. Because a goal was the reduction of parameters, tree was limited to three levels.*

bottom). This break point is almost identical to the Excellent SSD threshold for chloride of 40 mg/L (Table 8). The next level break was based on conductivity levels (>656 µS/cm) for attaining sites. The excellent SSD threshold is 738, so the lower cutoff of 656 may reflect a stream size effect or some other artifact in the data.

The General Use classification tree splits at a slightly higher chloride concentration (50 mg/L) which is influenced by the Excellent sites in the data set and sites with higher fish IBI scores. Lower branch splits (Figure 28, top) also include branches based on sodium on the "better" quality branches of the tree, and sulfate on the branches with more degraded sites. Overall, chloride appears to be the most important variable in this dataset with some additional explanatory power added by conductivity and then sodium and sulfate. ***[Variables to be included in overall regression and classification tree:*** *Chloride, Conductivity**]**.*

### *Organic Enrichment and Nutrient Parameters*

Because of the overlap between nutrient and organic enrichment parameters they are considered together in terms of these analyses. Given the extensive urban runoff and point source contributions in the study area, organic enrichment and nutrient parameters can be important limiting factors albeit in an indirect manner. The dataset also includes some historical data which could have elevated levels (e.g., ammonia, TKN) relative to more contemporary

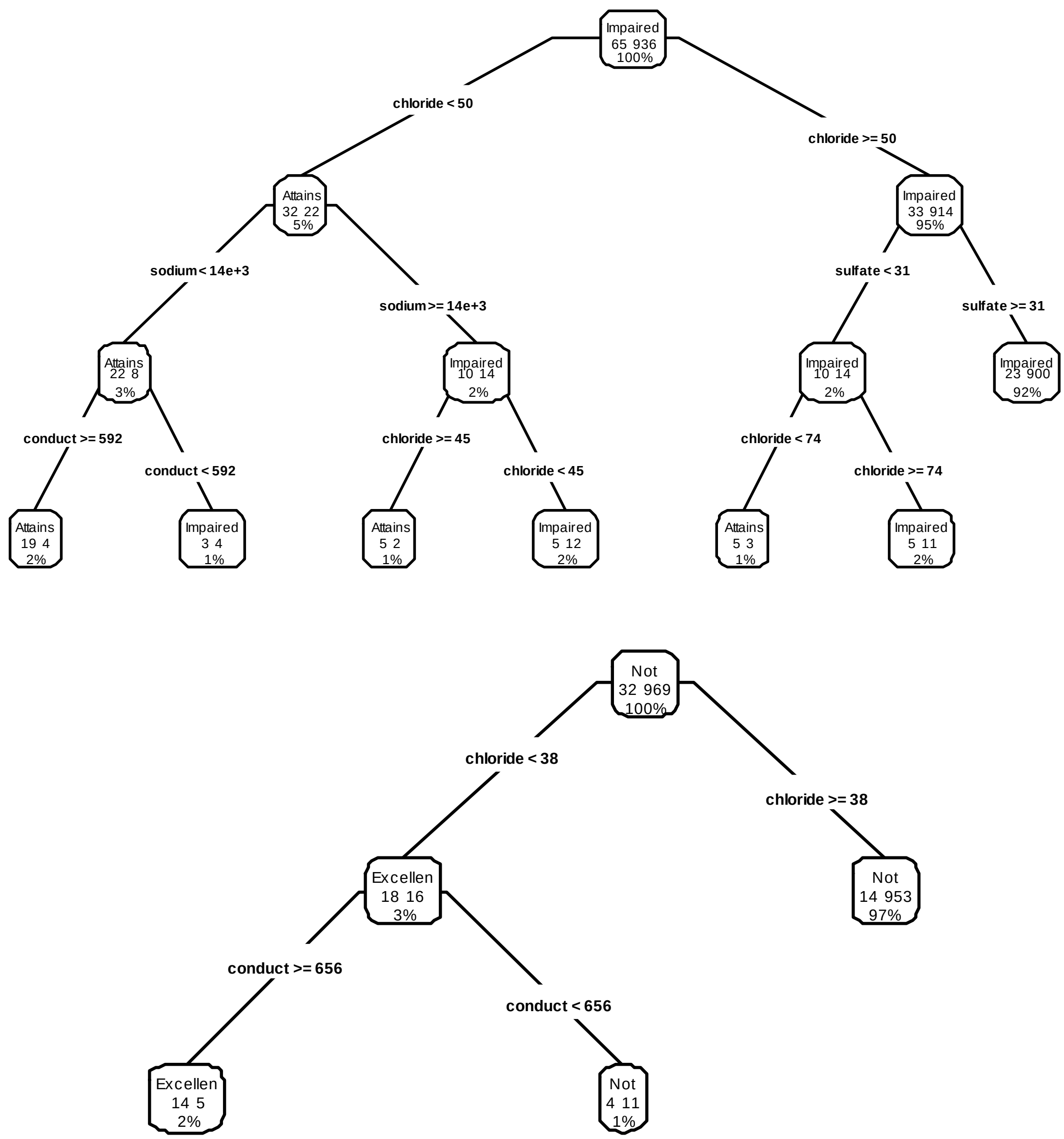



***Figure 28**. Results of two classification tree analysis with the attainment of the Illinois fIBI as the response variable (top) or attainment of the excellent narrative threshold (bottom) with ionic strength parameters (chloride, sodium, sulfate, potassium, total dissolved solid and conductivity) as explanatory values. Because a goal was the reduction of parameters, tree was limited to three levels.*

data. The nutrient and organic enrichment variables in the NE Illinois IPS are less well correlated, on average, than the habitat and ionic strength variables (Figure 29). These are based on grab samples and variables such as D.O. change over a 24 hour period which can add to the variability of grab sample data. Nitrate-N (usually dissolved) and total phosphorus (both total and dissolved) can be delivered to receiving streams and rivers via different pathways and over different time periods.

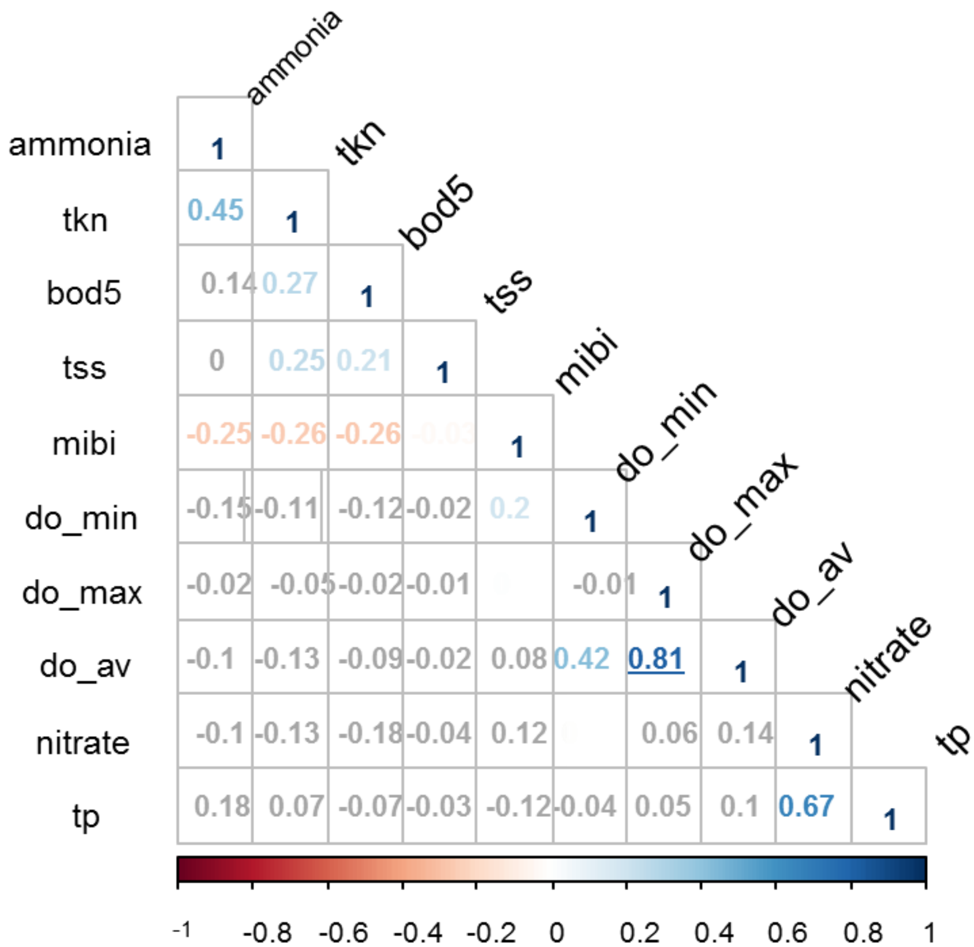



***Figure 29.*** *Matrix of Pearson correlation coefficients (r values) for nutrient and organic enrichment variables in the NE Illinois IPS study area. Highly correlated variables are clustered together. Positive correlations are blue and negative correlations red with color intensity greater with higher correlation coefficients; Coefficients > 0.7 are underlined.*

The first major split on the regression tree was on total ammonia-N concentrations (<u><</u>0.11 mg/L, Figure 30) which is only just above background levels. Sites with low ammonia-N had an average mIBI of 43 and sites with elevated ammonia-N an average mIBI of 33. Ammonia is a key variable associated with point source discharges and this parameter fits with the history of point sources in these watersheds. For sites with elevated ammonia-N, the next split is based on nitrate-N concentrations (<u>></u>1.8 mg/L) and could also be a marker for point source effluent or nonpoint source runoff. The next split in that branch is on total phosphorus (<u><</u>2.0 mg/L TP) which separates nodes with higher mIBIs (mean = 40) from those with a mean mIBI of 27 (>2.0 mg/L TP). On the furthest right side of the tree with the highest mIBI scores (means of 52 vs. 62), the higher scores were associated with lower TKN values (Figure 30). The lower branches on the trees generally have more uncertainty and although RPART (Therneau and Atkinson 1997) has ways of dealing with missing values, the incomplete nature of the variables among sites can contribute to some added variability in the branches. Figure 30 illustrates a classification tree based on the nutrient and organic enrichment variables that best explain attainment of the General Use (Good) mIBI (top) or the Excellent mIBI narrative (bottom). As with the regression tree for ammonia-N was responsible for the first split between attaining

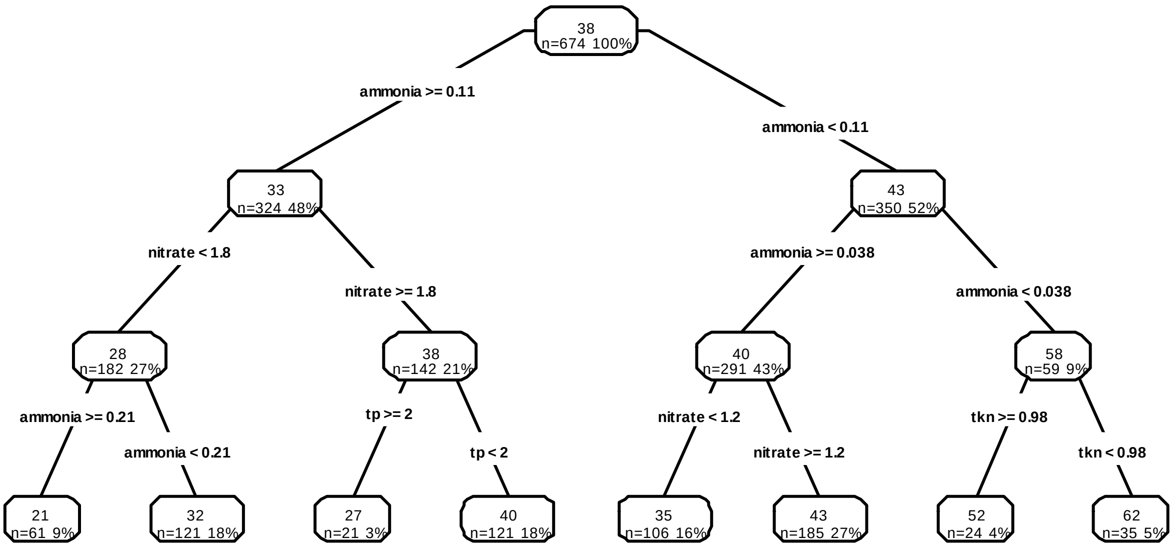



***Figure 30**. Results of a regression tree analysis with the Illinois mIBI as the response variable and organic enrichment and nutrient parameters (ammonia, TP, nitrate, BOD, D.O., etc.). Because a goal was the reduction of parameters, the regression tree was limited to three levels.*

and impaired sites and then a combination of ammonia-N, nitrate-N, and TP were key for other splits (Figure 31, top). The Excellent narrative classification was simpler tree with ammonia-N the key variable for all of the splits. Based on all three trees the most important variables are ammonia-N, nitrate-N, TP, and TKN. Other variables such as maximum and minimum D.O. are undoubtedly important, but would be better represented by continuous sampling results, better than the grab sample data that was used herein. ***[Variables to be included in overall regression and classification tree:*** *Ammonia, Nitrate, TKN, TP**].***

### *Suspended Materials and Sedimentation*

Total suspended solids and its analogs are frequently used as indicators of stormwater impacts, urban development, and other land use impacts. Suspended sediment is often considered as an indicator for the potential for siltation and other fine materials to be delivered to streams and rivers. The relationship between suspended sediment and aquatic life can be complex. Newcombe and Macdonald (1991) reported that concentration alone was a weak indicator of suspended sediment effects and found that a measure that included the concentration and duration of exposure was a better indicator. This matches our experience that shows individual grab samples, or averages or medians from a few samples are a weak predictors of biological condition. The QHEI substrate metric component scores are more strongly associated with biological condition and it is likely that continually elevated suspended sediments are associated with increased sedimentation and siltation and are likely mimicking a “Stress Index” as proposed by Newcombe and Macdonald (1991). Hence, the QHEI substrate metrics were included in the correlation and classification and regression tree analyses.

QHEI substrate related attributes were well correlated with one another (Figure 32) and TSS, VSS, and turbidity were correlated with one another. However, suspended variables were not

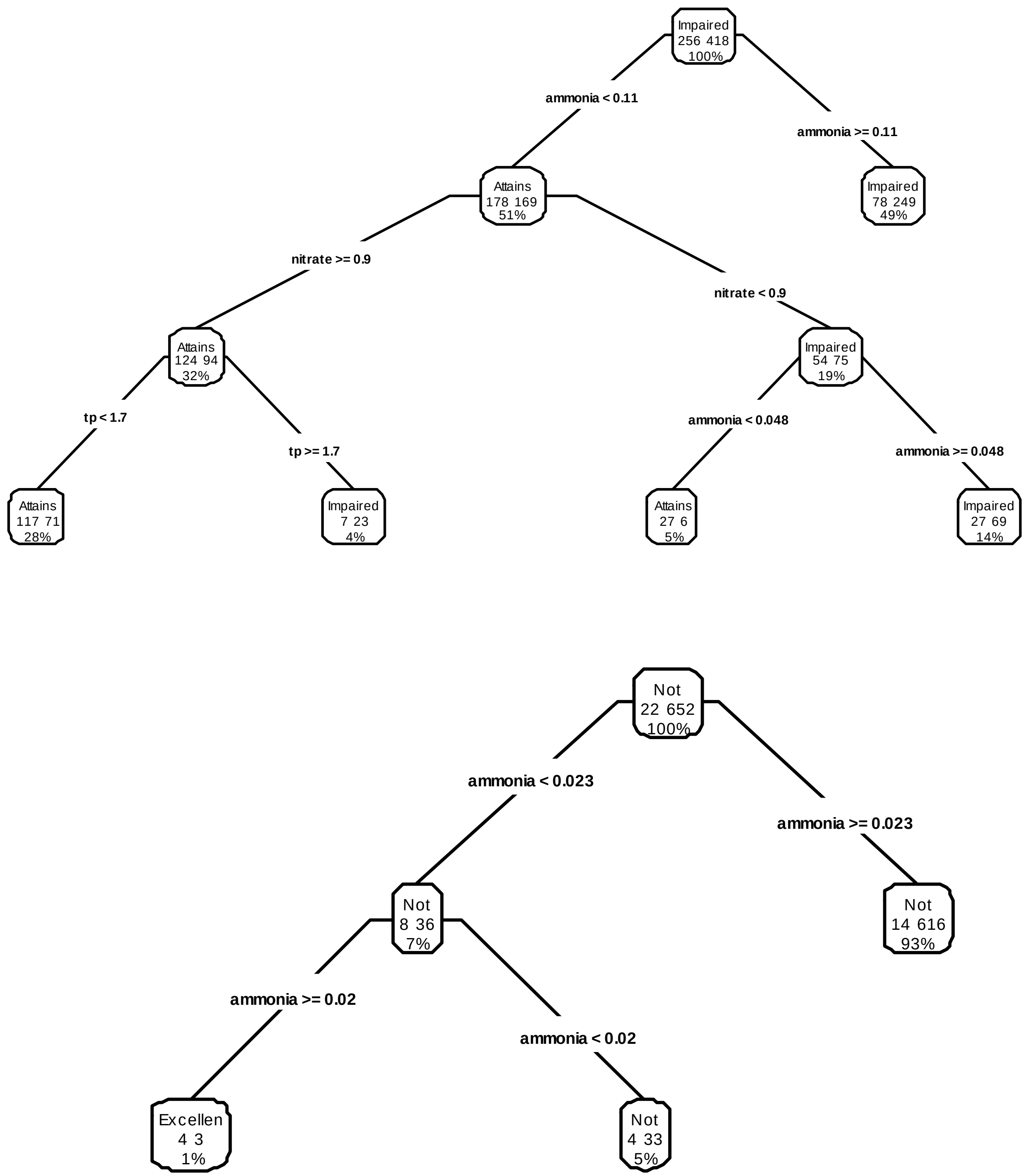



***Figure 31****. Results of two classification tree analysis with the attainment of the Illinois mIBI as the response variable (top) or attainment of the excellent narrative threshold for mIBI (bottom) with organic enrichment and nutrient parameters (ammonia, TP, nitrate, BOD, DO, etc.) as explanatory values. Because a goal was the reduction of parameters, tree was limited to three levels.*

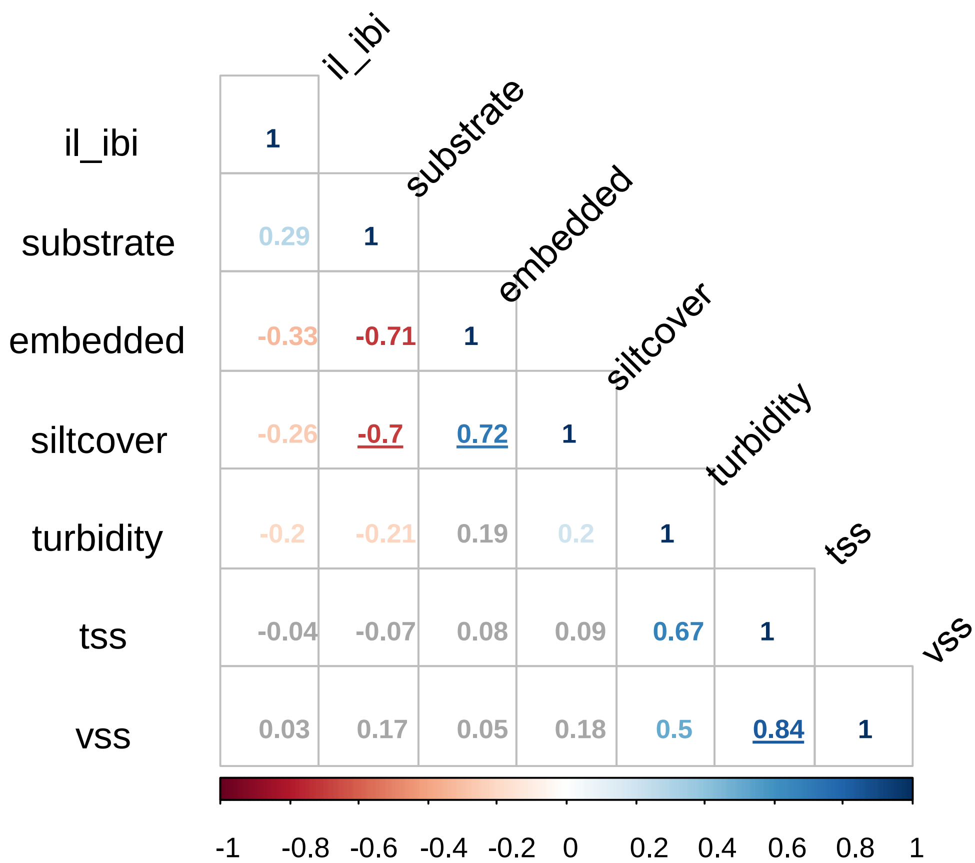



***Figure 32.*** *Matrix of Pearson correlation coefficients (r values) for suspended and bedded sediment variables in the NE Illinois IPS study area. Highly correlated variables are clustered together. Positive correlations are blue and negative correlations red with color intensity greater with higher correlation coefficients; Coefficients > 0.7 are underlined.*

strongly correlated with the QHEI substrate metrics. Measures of suspended sediment provide only brief snapshots and unless duration and magnitude are quantified it will be an inconsistent indicator of the effect of suspended sediments on biological condition.

Regression tree analyses (Figures 33 and 34) revealed the first split in the tree was on embeddedness with other splits on substrate score, turbidity, and VSS. Some of the variation in turbidity and VSS was due to its restricted availability, i.e., only collected in Lake Co. There was insufficient variation in the substrate scores to generate a classification tree for the attainment of the Excellent narrative IBI range. Suspended sediment is often used as a stormwater surrogate measure for sediment effects that may misclassify the nature of impacts associated with suspended materials. ***[Variables to be included in overall regression and classification tree:*** *Embeddedness, Substrate, Turbidity, VSS****]****.*

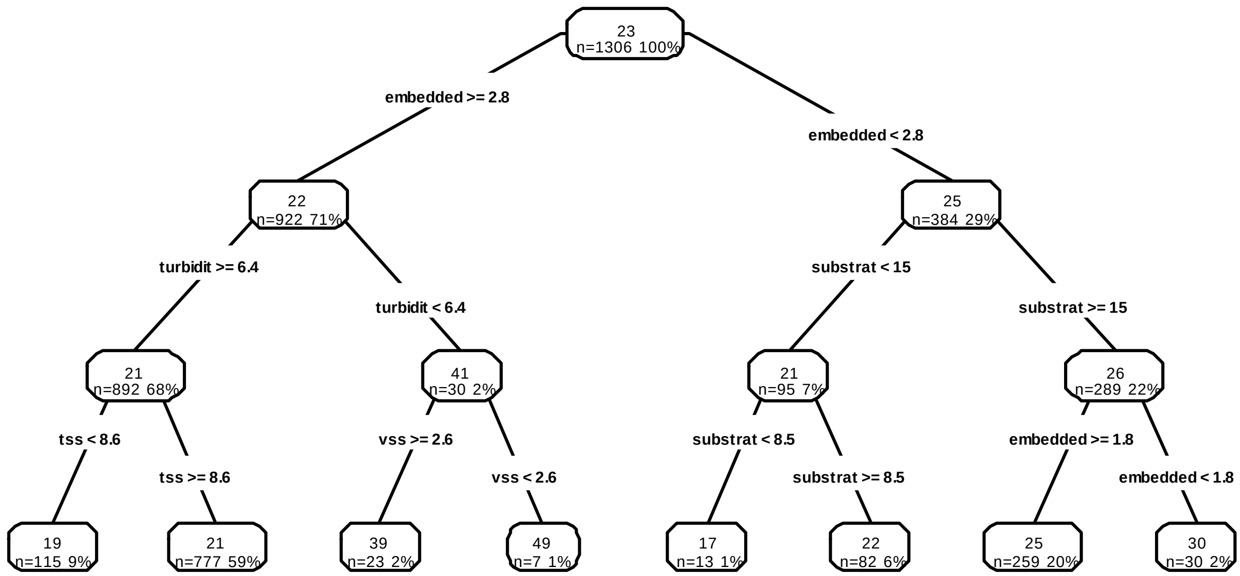



***Figure 34.*** *Results of a regression tree analysis with the Illinois fIBI as the response variable and suspended and bedded sediment parameters (TSS, turbidity, embeddedness score, substrate score, etc.). Because a goal was the reduction of parameters, tree was limited to three levels*.

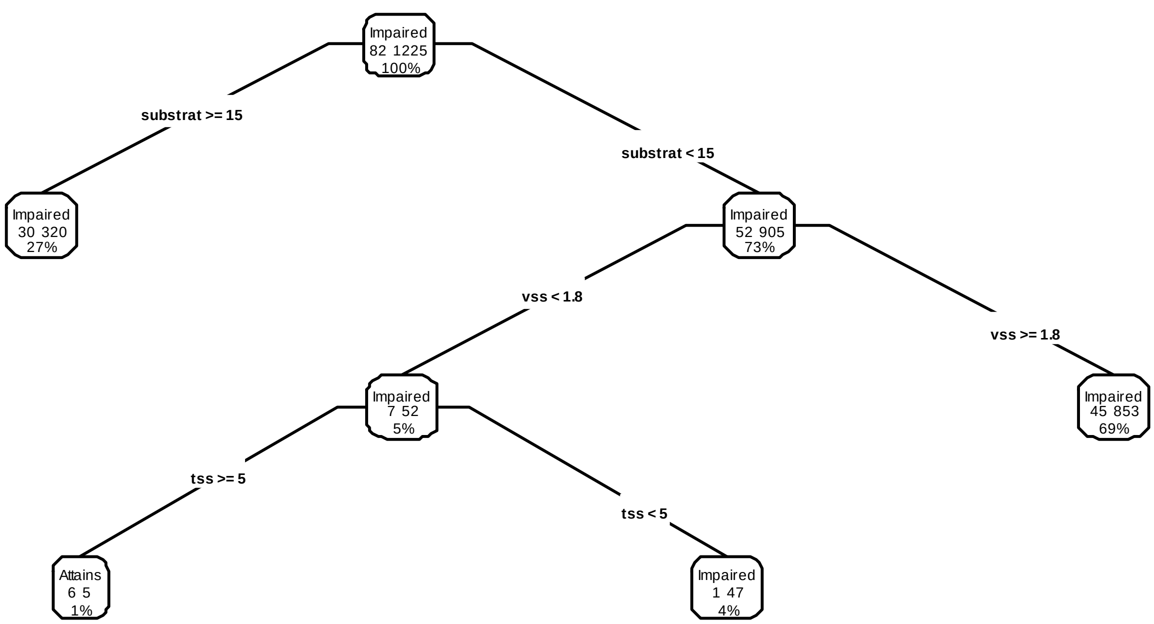



***Figure 33**. Results of a classification tree analysis with the attainment of the Illinois fIBI as the response variable with suspended and bedded sediment parameters (TSS, turbidity, embeddedness score, substrate score, etc.) as explanatory variables. Because a goal was the reduction of parameters, tree was limited to three levels.*

## Sediment Chemistry Parameters (PAHs and Metals)

Contaminated sediments are associated with aquatic life impairments in areas with moderate to high development (e.g., roads and other impervious surfaces) or historical industrial activity (manufacturing, mining). Sediment contaminants metals and PAHs are ubiquitous in urban and suburban watersheds.

### *Polycyclic Aromatic Hydrocarbon (PAH) Compounds*

Most PAH compounds were strongly correlated with one another and negatively correlated with the mIBI (Figure 35). This is understandable given that PAH compounds arise from similar sources in urbanized watersheds (asphalt, refined coal tar-based pavement sealcoat, fossil fuel combustion, tire degradates, etc.). The regression tree of the mIBI with PAH compounds (Figures 36 and 37) identified the first split (low average vs. higher average mIBI scores with Benzo(b)fluoranthene (<746 µg/kg) which is above the General Use SSD threshold of 520.8 µg/kg. There are no TEC or PEC consensus screening guidelines or the NOAA (2008) compendiums for this parameter. The General Use classification tree first splits (Figure 36) were on PAH compounds that categorize several levels of impairment, but the split that separated attaining sites from impaired sites was on Benzo(k)fluoranthene at 64 µg/kg, a level substantially lower than the General Use SSD threshold (207 µg/kg) and the LEL level (240 µg/kg) and well below the SEL level of 13,400 µg/kg, a level lower than the General Use SSD threshold (207 µg /kg) and the LEL level (240 µg/kg) and well below the SEL level of 13,400 µg/kg.

| | |
|---|---|
| P34203 | Acenaphthylene |
| P34208 | Acenaphthene |
| P34223 | Anthracene |
| P34233 | Benzo(b)fluoranthene |
| P34245 | Benzo(k)fluoranthene |
| P34250 | Benzo(a)pyrene |
| P34262 | Delta-BHC |
| P34323 | Chrysene |
| P34379 | Fluoranthene |
| P34384 | Fluorene |
| P34406 | Indeno(1,2,3-cd)pyrene |
| P34445 | Naphthalene |
| P34464 | Phenanthrene |
| P34472 | Pyrene |
| P34480 | Thallium |
| P34524 | Benzo(g,h,i)perylene |
| P34529 | Benzo[a]anthracene |
| P34559 | Dibenz(a,h)anthracene |

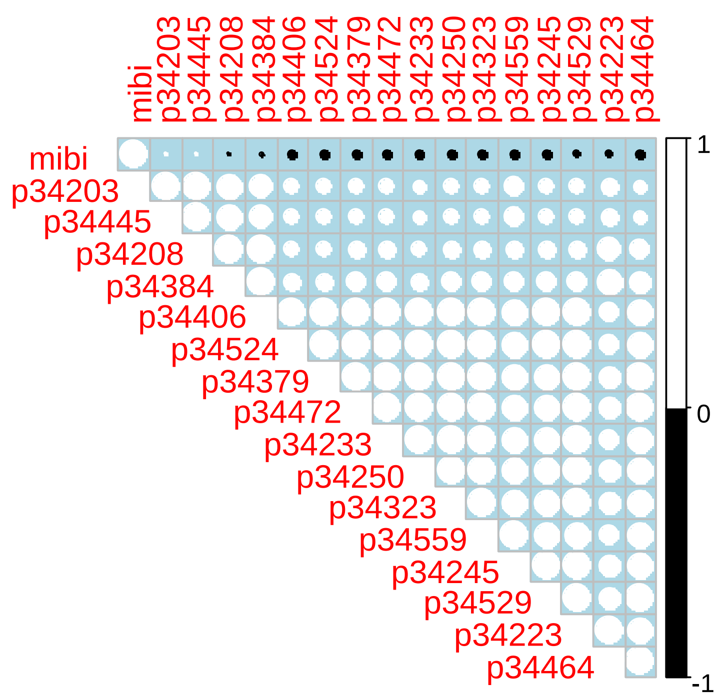



***Figure 35**. Matrix illustrating Pearson correlation coefficients for sediment PAH compound variables in the NE Illinois IPS study area. Highly correlated variables are clustered together. Positive correlations are white circles indicating higher correlation coefficients on left is key to parameter codes.*

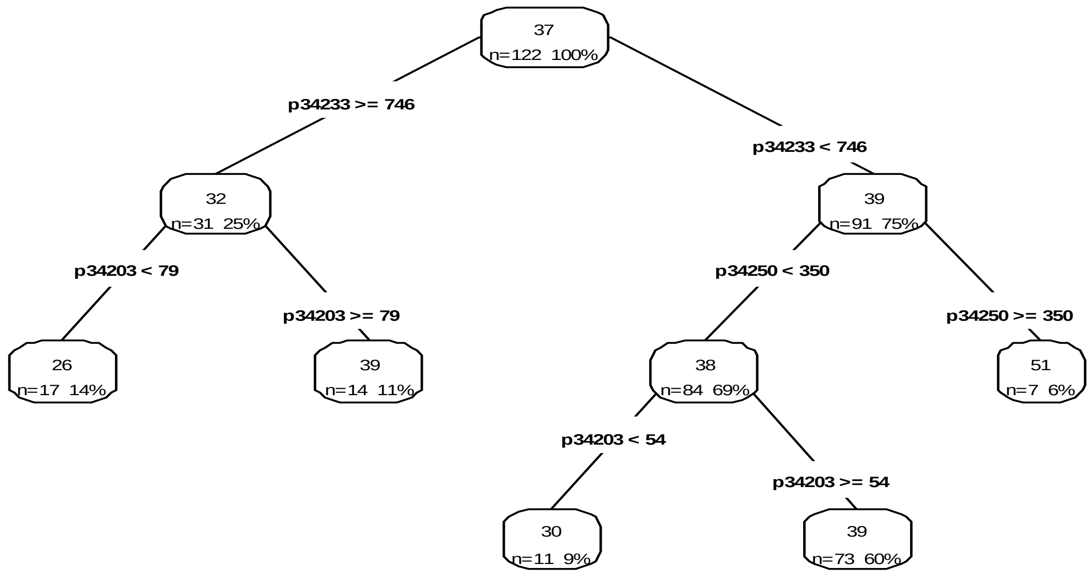



*Figure 36. Results of a regression tree analysis with the mIBI as the response variable and sediment PAH parameters and explanatory variables (see Fig. 35). Because a goal was the reduction of parameters, tree was limited to three levels.*

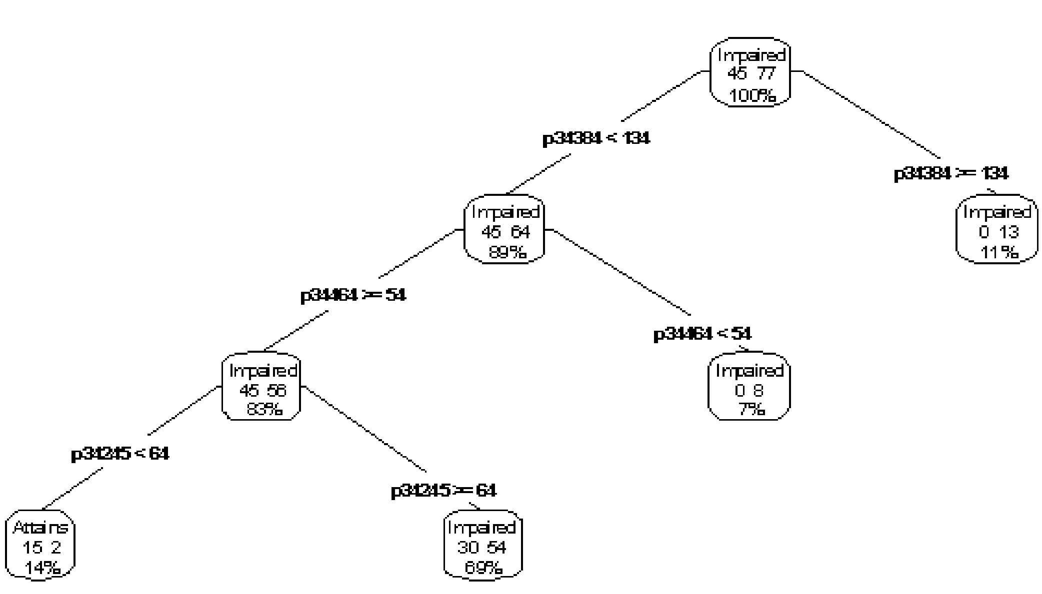



*Figure 37. Results of a regression tree analysis with the mIBI as the response variable and sediment PAH parameters and explanatory variables (see Fig. 35). Because a goal was the reduction of parameters, tree was limited to three levels.*

These analyses focused on individual PAH compounds. Future analyses should consider groupings of parameters (e.g., total PAHs or low or high molecular weight compounds per CEQC 2011). Research has identified that toxicity from PAHs may act in a cumulative or synergistic fashion (Verrhiest et al. 2001) and dependent on their availability in the water column (Crunkilton and DeVita 1997). The regression and classification tree did not point to any specific PAH compounds as being strongly causative by themselves and some of the splits occurred at levels below the General Use SSD thresholds and TEC/TEL/LEL concentrations. The lack of sufficient PAH data at sites with Excellent mIBI scores likely compromised a stronger response gradient in the macroinvertebrate sensitive taxa SSD response, i.e., insufficient data relating to determine where some sensitive taxa may decline or are absent. Additional data from Excellent performing reference sites will likely strengthen these analyses. ***[Variables to be included in overall regression and classification tree:*** *Benzo(b)fluoranthene, Acenaphthylene, Benzo(a)pyrene, Benzo(k)fluoranthene****].***

**Heavy Metals in Sediment**

Most of the sediment metal parameters were strongly correlated with another, but there were some exceptions (Figure 38). The metals that were the most strongly correlated (negatively) with the mIBI included copper, aluminum, and zinc, each of which are associated with runoff from paved surfaces. In the regression tree analysis the first split was on copper at a level of 21 mg/kg which is lower than the General Use (Good) SSD threshold of 29.8 mg/kg (Table 9), but the split does include mIBI scores above the Good mIBI threshold of 41.8. For sites that had an average mIBI of 44, the split that separated these sites from ones with an average mIBI of 30 (Fair) was zinc at a concentration of <190 mg/kg which is above the General Use (Good) SSD threshold of 100 mg/kg and the TEC guideline of 121 mg/kg. However, it is well below the PEC guideline of 459 mg/kg. Splits on the right side of the tree that distinguished among sites with Good or Excellent mIBI scores were at levels below the Good mIBI SSD threshold. These may well be covariates with other parameter levels that are associated with higher levels of biological condition.

The classification tree for sediment metals (Figure 39) with General Use (Good) attainment as the response variable, split first on silver with the subsequent separation of attainment vs. impairment based on zinc (182 mg/kg) and copper (20 mg/kg). A split on barium (38 mg/kg) was not meaningful because the impaired group split on the basis of having lower barium. Unlike PAHs, there was sufficient metals data to generate a classification tree analysis for the Excellent narrative category. The first split in the tree was on silver, but the Excellent split at the next level was on copper (23 mg/kg). Silver and barium do not have any literature based screening guidelines and the values that were associated with the splits in the regression and classification trees are below the Good SSD thresholds which suggests that the splits may be surrogates for other stressors such as sedimentation or other influences of urban land use. ***[Variables to be included in overall regression and classification tree:*** *Copper, Barium, Silver, Zinc****].***

| | |
|---|---|
| P1003 | Arsenic |
| P1008 | Barium |
| P1013 | Beryllium |
| P1028 | Cadmium |
| P1029 | Chromium |
| P1038 | Cobalt |
| P1043 | Copper |
| P1052 | Lead |
| P1053 | Manganese |
| P1068 | Nickel |
| P1078 | Silver |
| P1083 | Strontium |
| P1093 | Zinc |
| P1108 | Aluminum |
| P1170 | Iron |

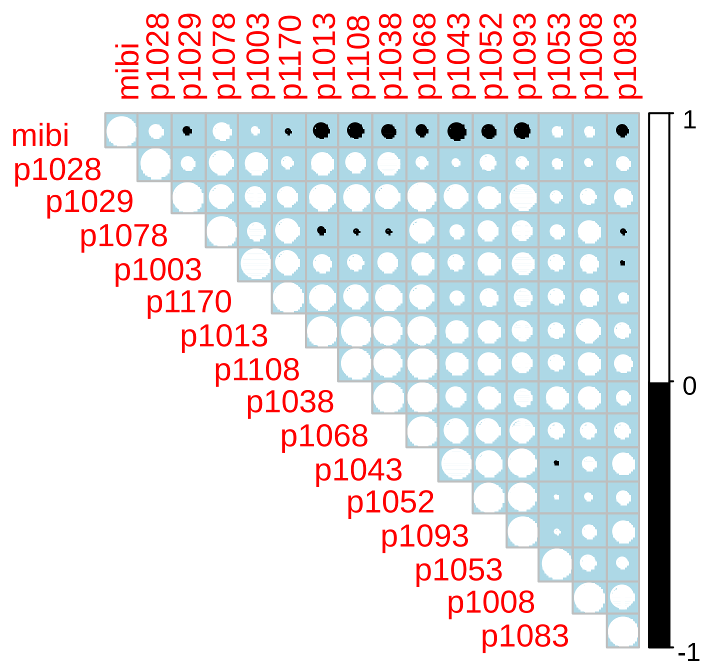



***Figure 38***. *Matrix illustrating Pearson correlation coefficients for sediment metal compounds in the NE Illinois IPS study area. Highly correlated variables are clustered together. Positive correlations are white and negative correlations are black with circle sizes indicating higher correlation coefficients.*

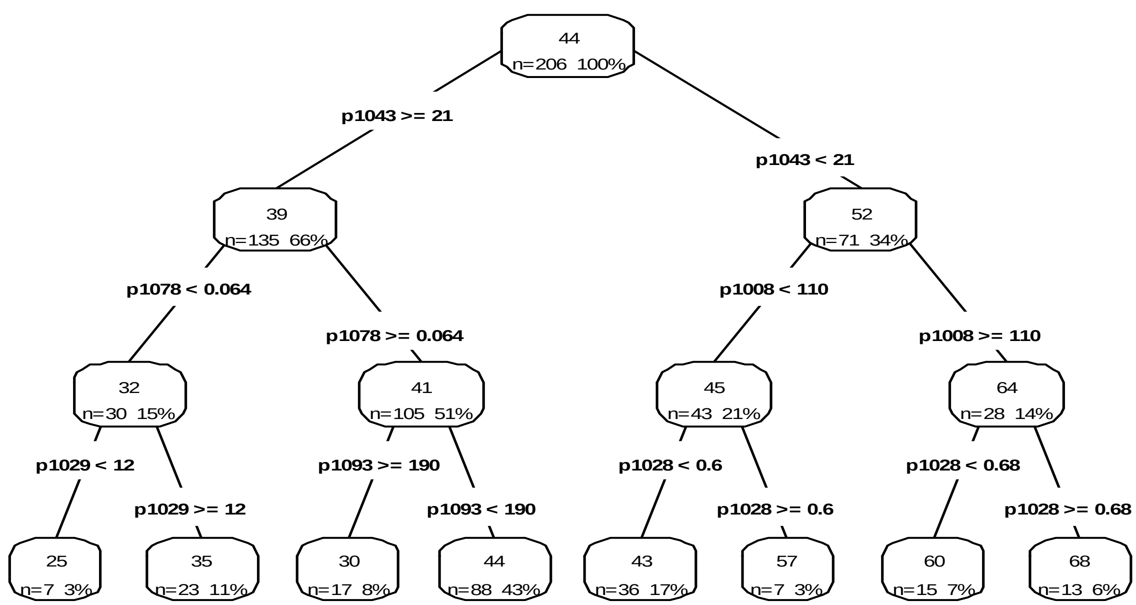
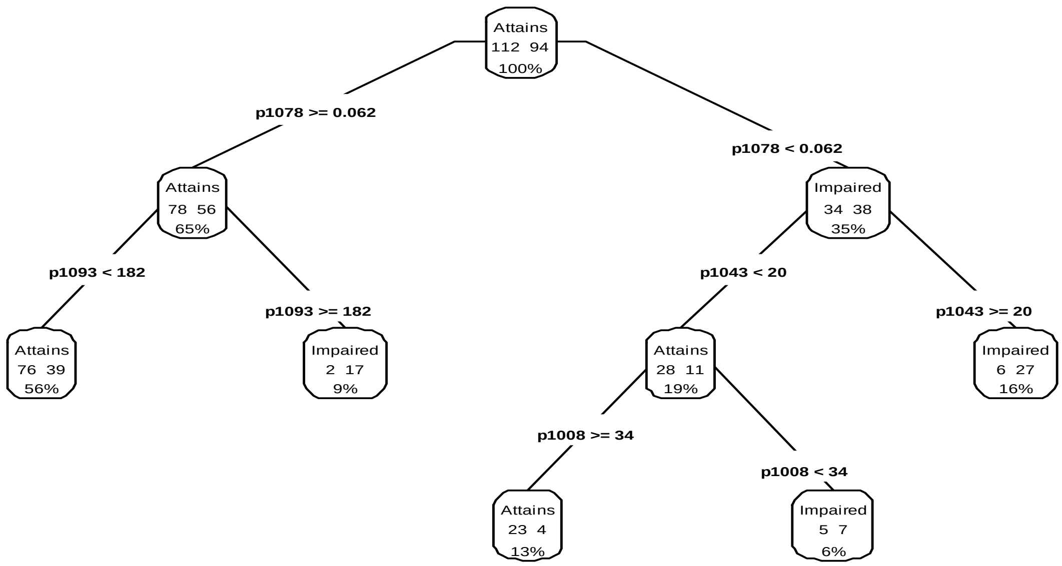



***Figure 39***. *Results of a regression tree analysis with the mIBI as the response variable and sediment metal parameters (copper, cadmium, zinc, etc.). Because a goal was the reduction of parameters, tree was limited to three levels.*



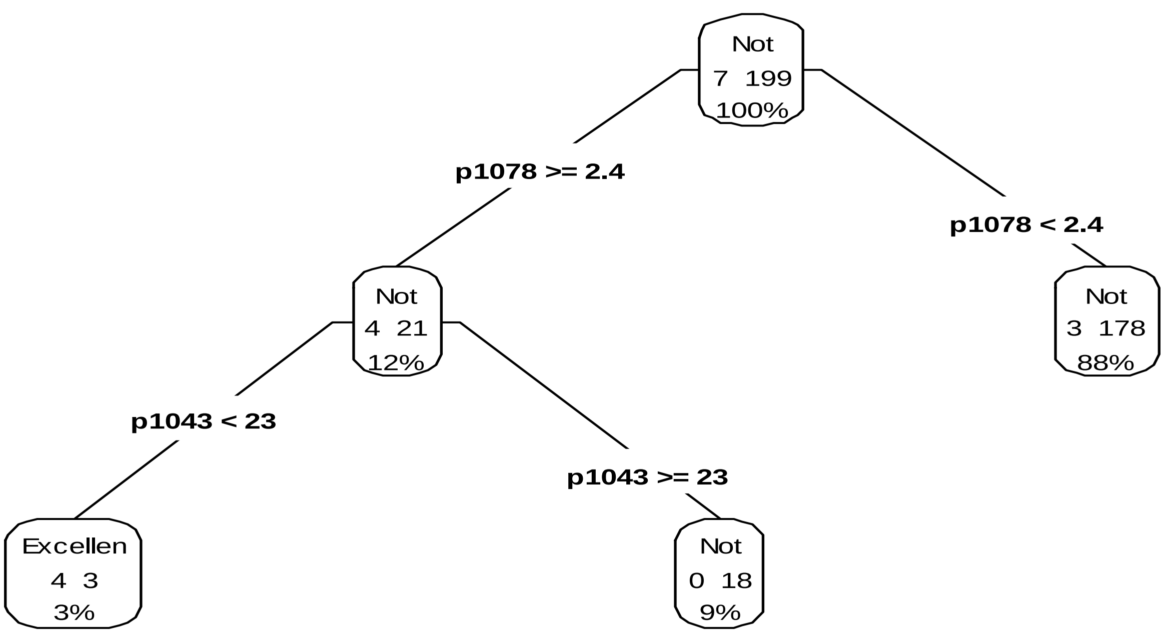




***Figure 40**. Results of two classification tree analysis with the mIBI as the response variable (top) or the excellent mIBI narrative (bottom) and sediment metal parameters (copper, cadmium, zinc, etc.). Because a goal was the reduction of parameters, tree was limited to three levels.*

## Land Use

As described in Chapter 3, land use is strongly associated with aquatic life condition with increasing developed urban and impervious land surfaces being limiting to aquatic assemblages (Schueler 2004). A series of CMAP and NLCD land use variables (agricultural land, grassland, forested land, wetlands, developed land, urban land, heavy urban land, and impervious cover) were calculated at the HUC12 watershed scale, within a 30 meter spatial buffer, and within a 500 meter spatial buffer, at the HUC12 watershed scale and clipped spatial buffers at the site scale (reduced to drainage area only).

Figure 41 is an illustration of the correlation coefficients (Pearson) between each of the land use variables in the NE Illinois IPS study area. The table was constructed using a hierarchical clustering procedure to group similarly correlated variables together. For the most part, each of the variables (agricultural, urban, etc.) across all scales (i.e., watershed vs. 30m spatial buffer, etc.) were strongly correlated and tended to cluster together (Figure 41). Developed land uses clustered together as did the natural land use categories. The mIBI was negatively correlated with all of the developed land use variables, but more strongly so with those at the watershed scale ('WS', '50' variables) than at the spatial buffer scale ('30', '3c' variables), but was only weakly correlated with natural land uses (e.g., forested, wetland, grassland, etc.).

The first split in the regression tree of the mIBI (Figure 41) vs. land use parameters was on developed land uses that split between higher and lower average mIBI scores, but some mIBI scores above the General Use (Good) mIBI biocriterion were on both sides of the split (Figure 42). The breakpoint was a high of 86% developed land use. The sites on the left side of the tree that had higher mIBIs (mean = 43) also had a higher proportion of natural land uses at the watershed scale (<7.4%) and wetland land cover in the 500m spatial buffer >2%. Sites on the left side of the tree with lower mIBIs had lower amounts of natural land and wetlands and more heavy urban land use in the 500m spatial buffer. The second split on the right side (higher mIBI scores) side of the tree was based on a higher fraction of agricultural land in the watershed (≥62%) which corresponded to the location of reference sites in the watershed. The highest mIBIs (mean = 65) corresponded to the highest levels of forested land (>1.6%).

The classification analysis with the General Use (Good) attainment of the mIBI as the response variable (Figure 43) showed the same first split as the regression tree at 86% of the watershed in developed land uses with most attaining sites on the left side which represented the lower developed land use side of the tree. For watersheds with a higher percentage of developed land use (right side of tree), the only sites that attained the General Use were the sites with a higher proportion of local wetlands. ***[Variables to be included in overall regression and classification tree:*** *Development – WS, Agriculture – WS, Natural – WS, Forest – WS, Wetland – 500m* ***]***.

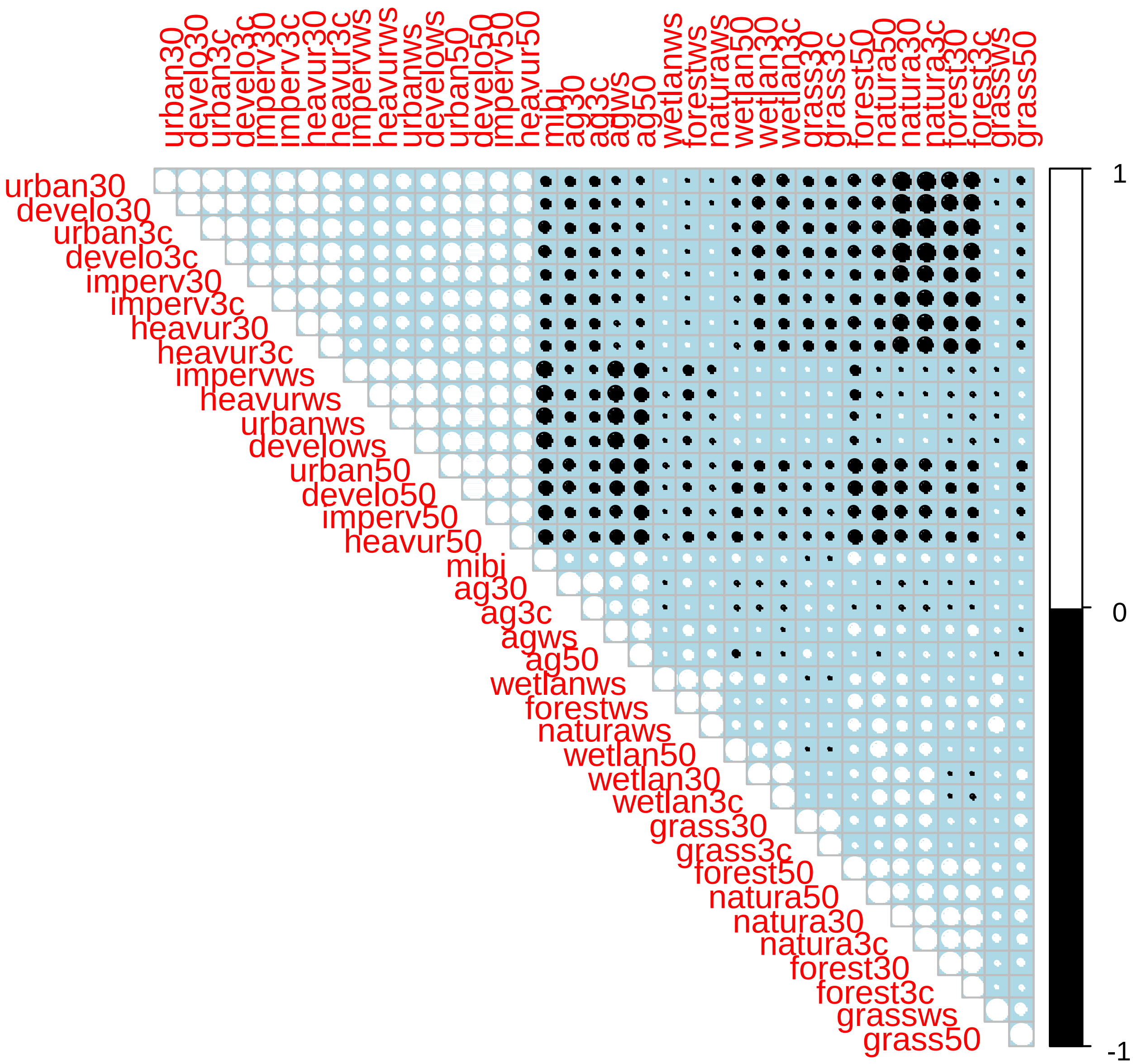



***Figure 41***. *Matrix illustrating Pearson correlation coefficients for land use variables in the NE Illinois IPS study area. Highly correlated variables are clustered together. Positive correlations are white and negative correlations black with large size circles indicating higher correlation coefficients; watershed scale land use variables end with 'WS', land use variables in a 30m buffer at a watershed scale end in '30'; land use variables in a clipped 30m buffer end in '3C' and land use variables in a 500m buffer end in '50'.*

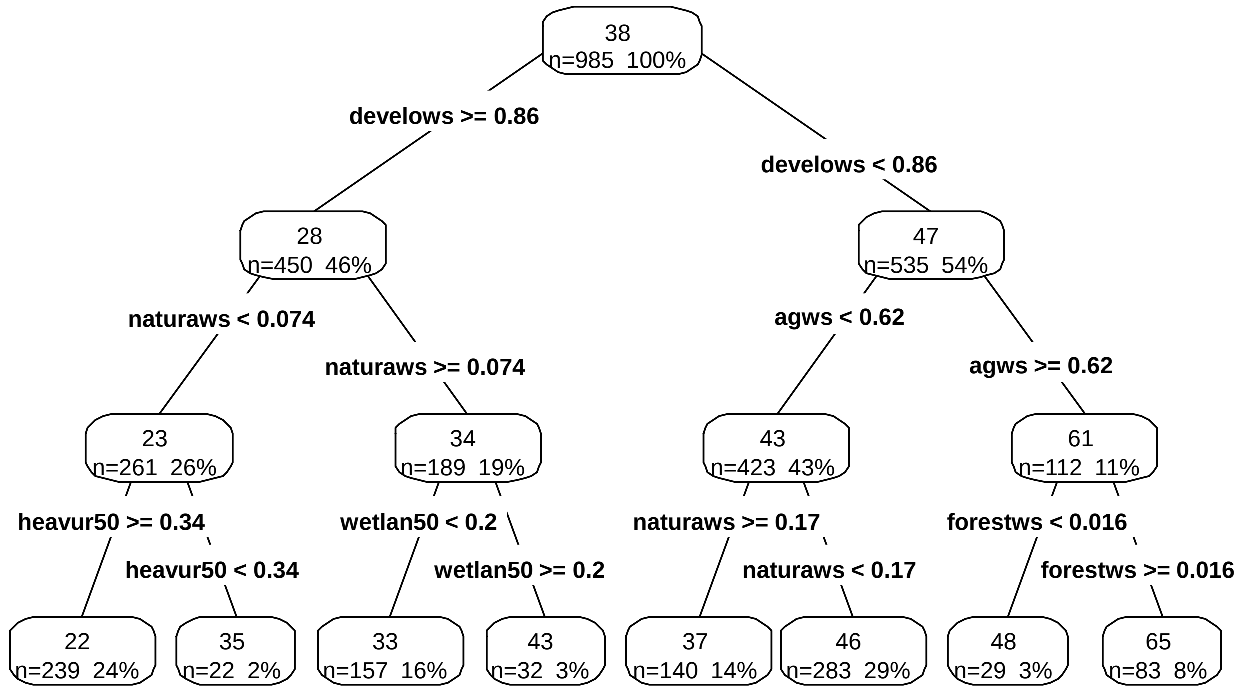



***Figure 42.*** *Results of a regression tree analysis with the mIBI as the response variable and land use parameters as explanatory variables. Because a goal was the reduction of parameters, tree was limited to three levels.*

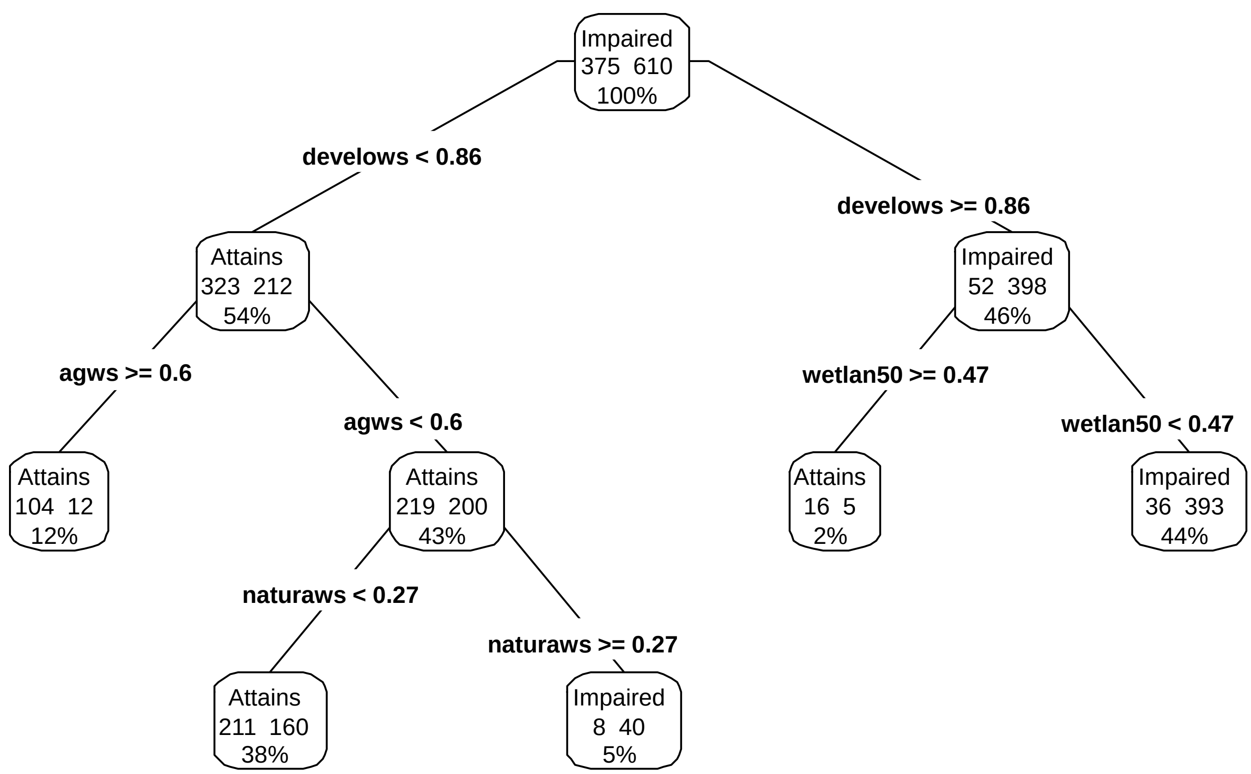



***Figure 43****. Results of classification tree analysis with the attainment of the Illinois macroinvertebrate mIBI as the response variable with land use parameters as explanatory values. Because a goal was the reduction of parameters, tree was limited to three levels.*

## Water Column Variables

### *Metals and Toxics*

Excluding ammonia, the most commonly occurring "toxic" compounds in the water column of streams and rivers are heavy metal compounds such as copper, cadmium, lead and zinc. The compounds formerly occurred at much higher concentrations in streams and rivers prior to controls mandated by the CWA, the development of water quality criteria for these pollutants, and the issuance of water quality-based effluent limits (WQBELs) in NPDES permits. Metals have been associated with chronic impacts such as elevated anomalies on fish and other aquatic life (Noga 2000) and their reduction in recent years in the water column is associated with improved biological assemblages.

Commonly occurring metals such as copper, cadmium, and zinc are highly correlated with one another (Figure 44) and are more likely to be associated with localized impairments than other metals such as iron, aluminum, and manganese which are frequently more benign in their effects. The regression tree analyses (Figure 45) suggest that aquatic life is more affected by the most toxic metals (lead, cadmium, and zinc) which comprise the split point for the General Use (Good) threshold.

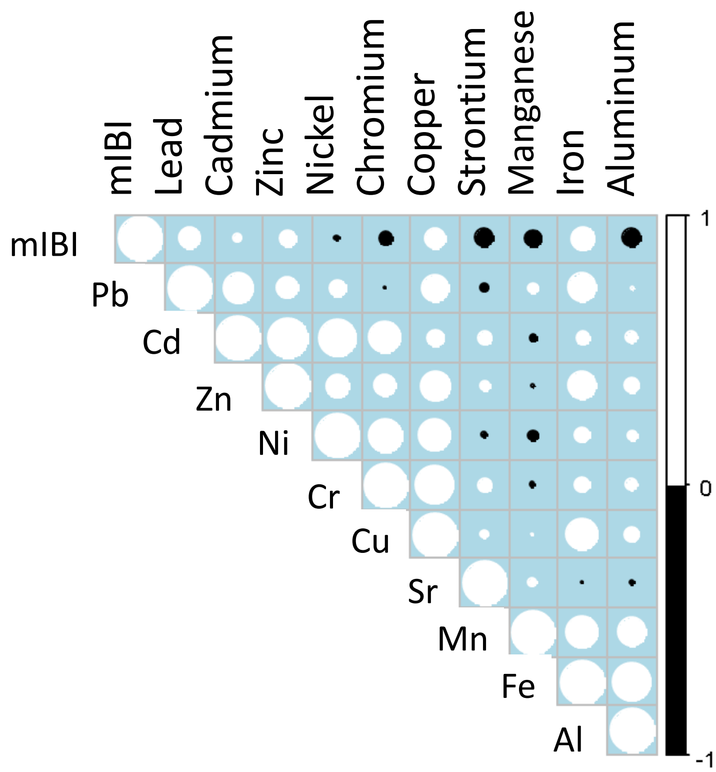



***Figure 44**. Matrix of Pearson correlation coefficients (r values) for water column metal variables in the NE Illinois IPS study area. Highly correlated variables are clustered together. Positive correlations are white and negative correlations are white with size of the circle indicating higher correlation coefficients.*

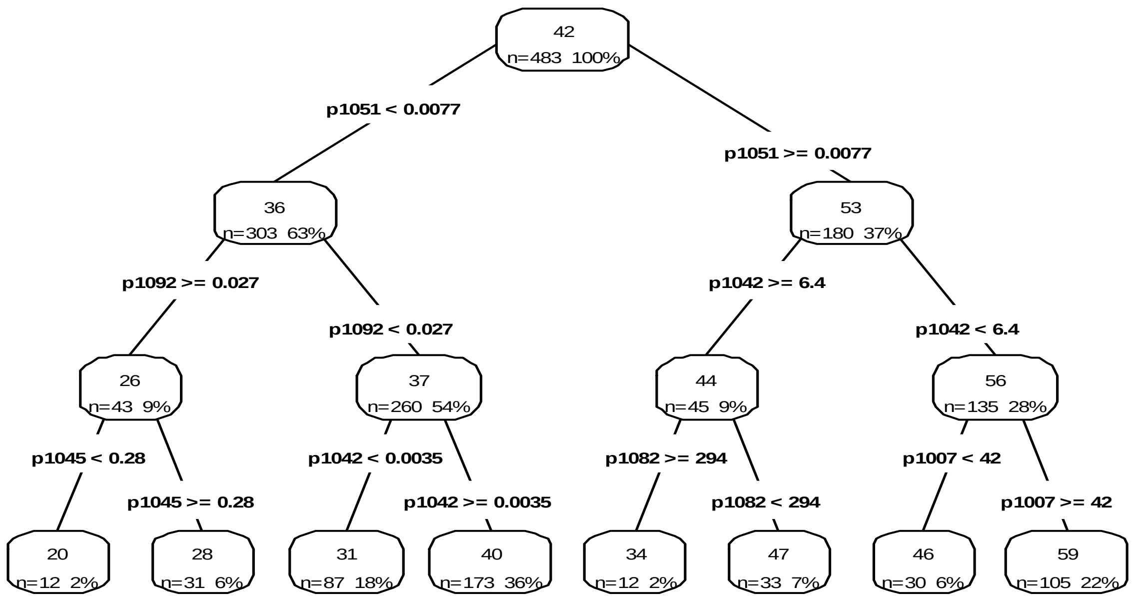


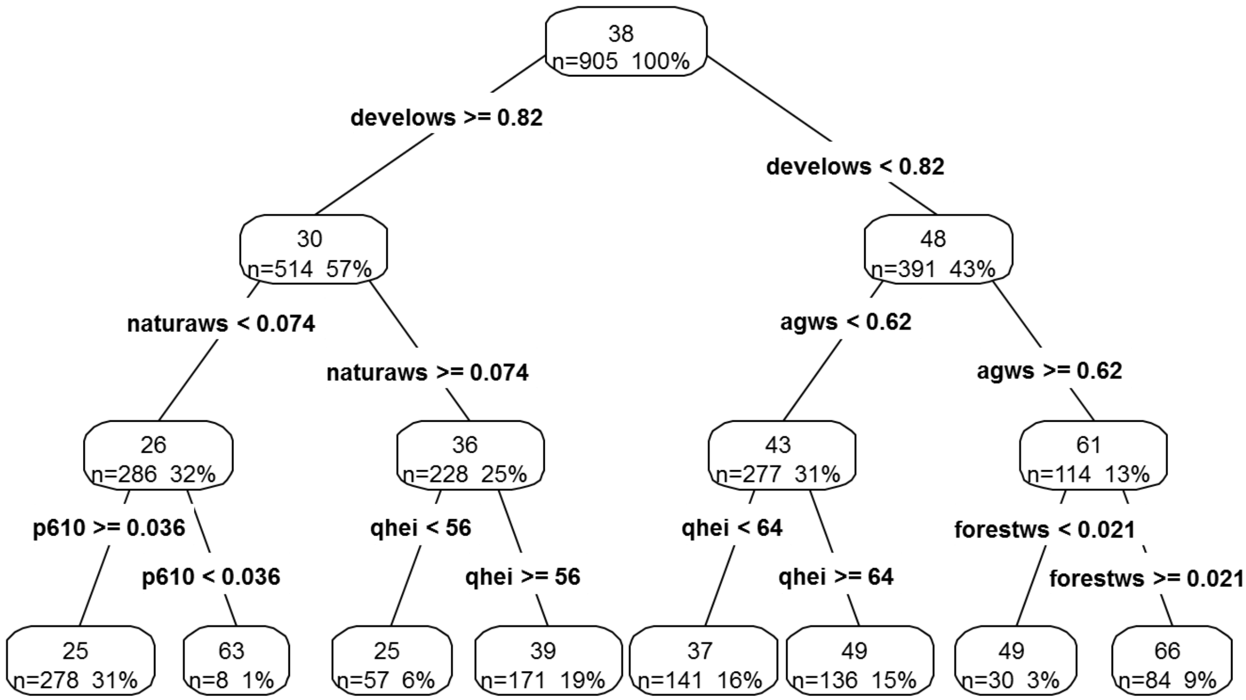

***Figure 45**. Results of a regression tree analysis with the mIBI as the response variable and water column metal parameters as explanatory variables. Because a goal was the reduction of parameters, tree was limited to three levels.*

## All Key Variables

On the basis of regression and classification tree analyses about stressor importance a group of selected stressor metrics were compared to one another in summary regression and classification tree analyses. Because the stressor relationships varied somewhat between fish and macroinvertebrates a series of analyses was performed with both assemblages, aquatic life use attainment status, and Excellent score performance based on the fIBI and mIBI.

In both the mIBI and the fIBI regression tree analyses (Figure 46), developed land use at the watershed scale was the origin of the first split in the trees. The second level of branches split on natural and agricultural land uses at the watershed scale, and the third level included splits on QHEI, ammonia-N, and forested land uses. The highest mIBIs were associated with sites <82% developed land use, >62% agricultural land use, and >2% forested land use. For the fIBI developed land use splits were common and QHEI was important in the second level. The sites with Excellent fIBIs had <10% developed land uses and QHEI scores >84 which are similar to the SSD threshold analysis results (Table 10).

The General Use classification tree (Figure 47) splits for macroinvertebrates were similar to the regression tree with the first splits on developed land uses at the watershed scale and subsequent splits on QHEI and ammonia-N. For fish, the first split changed to agricultural land use with more frequent fIBI attainment at sites with higher agricultural land uses which was



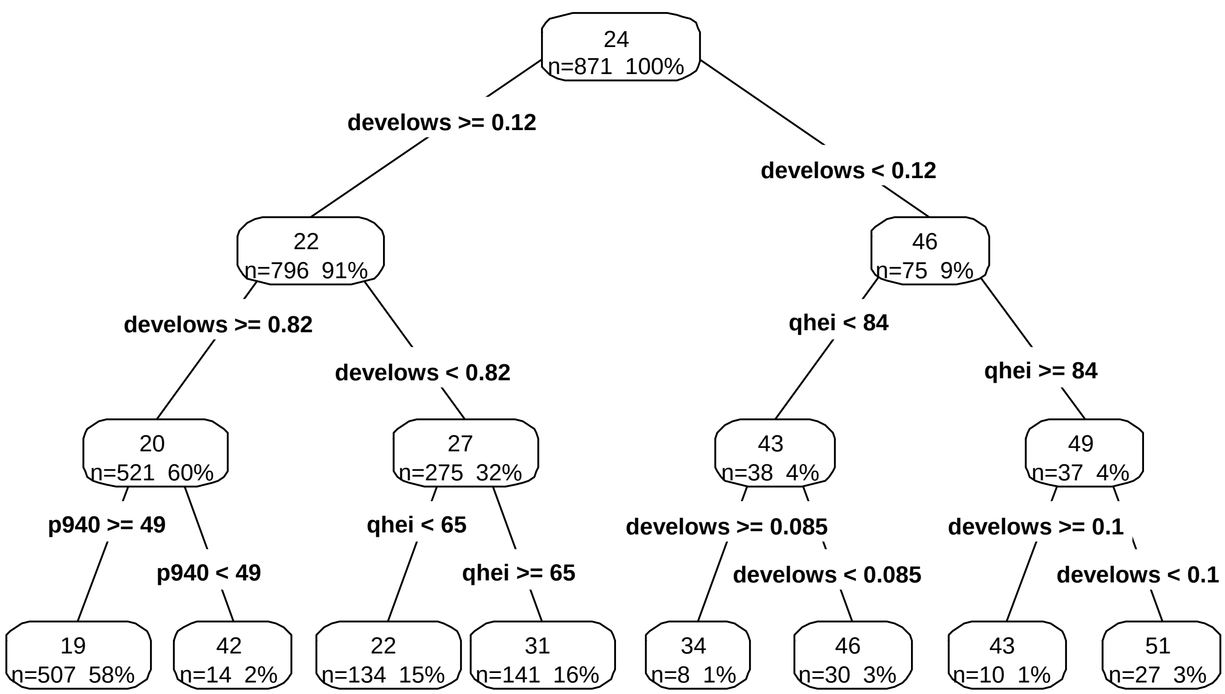




***Figure 46**. Results of a regression tree analysis with the mIBI as the response variable (top)) and the fish IBI as the response variable (bottom) and all key variables as explanatory variables. Because a goal was the reduction of parameters, tree was limited to three levels*

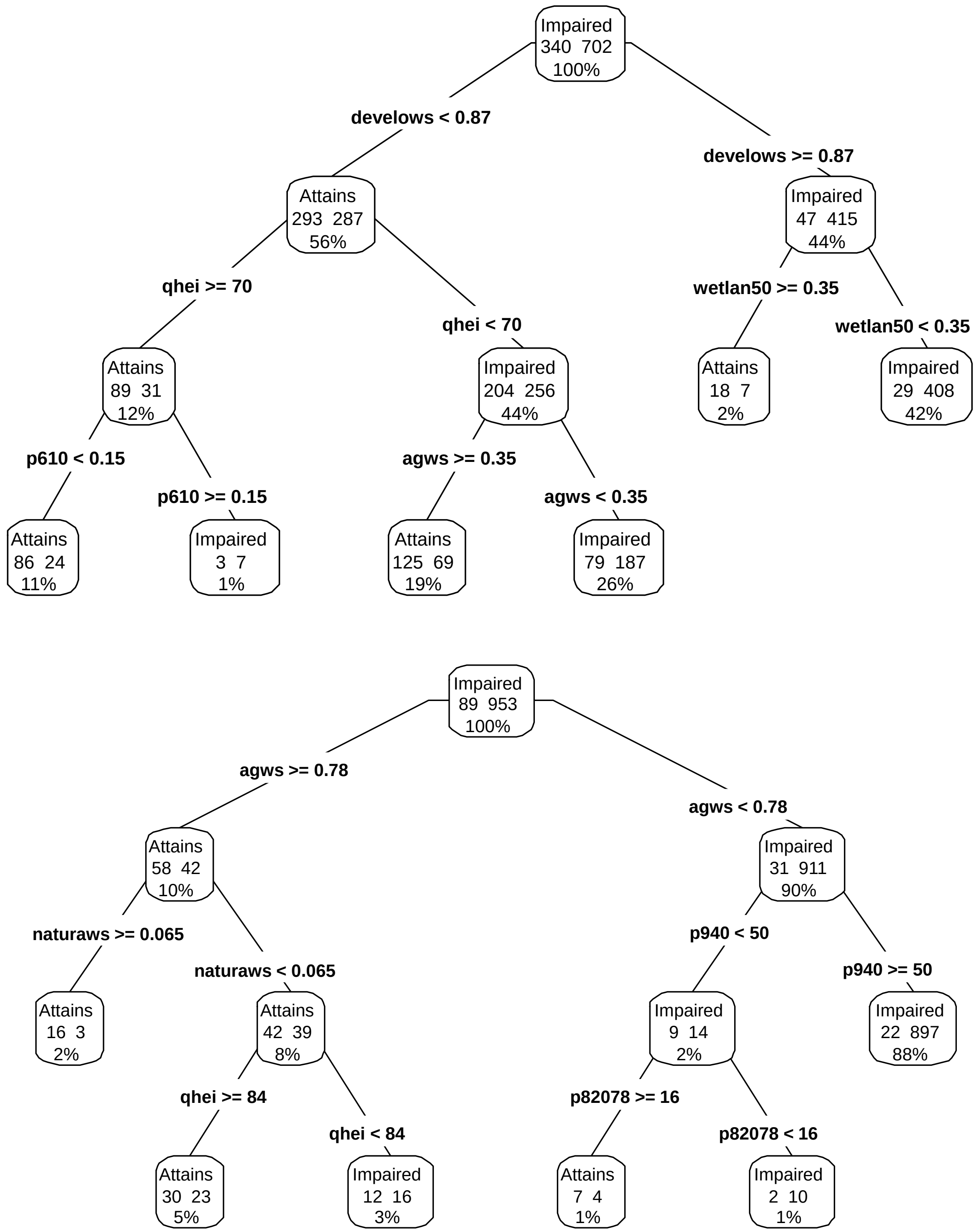



***Figure 47**. Results of a classification tree analysis with the mIBI General Use threshold as the response variable (top) and fish IBI (bottom) and key stressor parameters from each stressor category as explanatory variables. Because a goal was the reduction of parameters, tree was limited to three levels.*

likely the result of more consistently better performing sites occurring mostly in the outlying watersheds with higher proportions of agricultural land uses. The Excellent narrative range classification tree (Figure 48) for macroinvertebrates was simpler with Excellent sites split by low watershed development (<14%) and high QHEIs (>91). For fish, the most frequent Excellent sites were split by QHEI scores ≥84 and <9.2% developed land uses. Where the QHEI was <84, then Excellent sites were split by TP values of <0.08 mg/L, which is similar to the SSD threshold.

The results of the regression and classification analyses were not unexpected in that key variables including developed land uses, QHEI, agricultural land use (as an indicator of Good conditions), and then ammonia-N, total P, and chloride were the key stressor variables in certain circumstances. Land use can act as a surrogate for stressor relationships that are weakened by missing data for key pollutants and because it integrates multiple and different stressor modes of action. Habitat is of known importance and high QHEI scores are needed at a minimum, for Good and Excellent biological performance in a developed landscape. In Ohio, for example, in less developed watersheds, Exceptional biological performance is associated with slightly lower QHEI scores (>75), although the best performing sites generally have QHEI scores >80. Neither sediment PAHs nor sediment metals showed as splits, however that data is sparser than other stressor variables and was especially lacking at sites with Excellent mIBI and fIBI scores. This analysis used the entire dataset to obtain a first approximation of key stressors and predictive variables in the NE Illinois IPS study area and to provide variables for inclusion in the Random Forest analyses. One recent study of urban watersheds across the U.S. (Bradley et al. 2023) shows that the number of chemical stressors in urban watersheds are being grossly under-counted and which are almost certainly contributing to the "urban stream syndrome"[12] (Walsh et al. (2005).

## FIT Analysis Results

The FIT (goodness-of-fit) statistic was developed to measure the strength of the relationships between the observed parameter ranks and predicted ranks based on the species or taxa richness of parameter sensitive species, i.e., the SSDs. Table 12 contains FIT scores for key IPS parameters ranked by FIT score with lower scores indicating a better goodness-of-fit across a range of stressor categories. Parameters with the "tightest" fit include land use and habitat variables (Table 12) along with dissolved materials parameters (chloride, conductivity) and organic/nutrient parameters (total phosphorus, minimum dissolved oxygen and TKN.

When causes of impairment are assigned, it inherently carries a relative degree of certainty or uncertainty about that assignment. The FIT weighting score influences the categories of narrative condition (i.e., Very Poor, Poor, or Fair) in which each cause of impairment is placed. Each stressor is ranked from 0.1 (excellent quality) to 10 (very poor quality) based on the respective relationships with the number of stressor-sensitive fish species or macroinvertebrate taxa as the response variable with a particular stressor via the SSD analysis. Where the association is very strong (i.e., FIT value <0.10) it means there were very few outliers and presumably a better power of prediction. The weighting factor is 1 and stressors that scored as

[12] The term "urban stream syndrome" describes consistently observed ecological degradation of streams draining urban land.

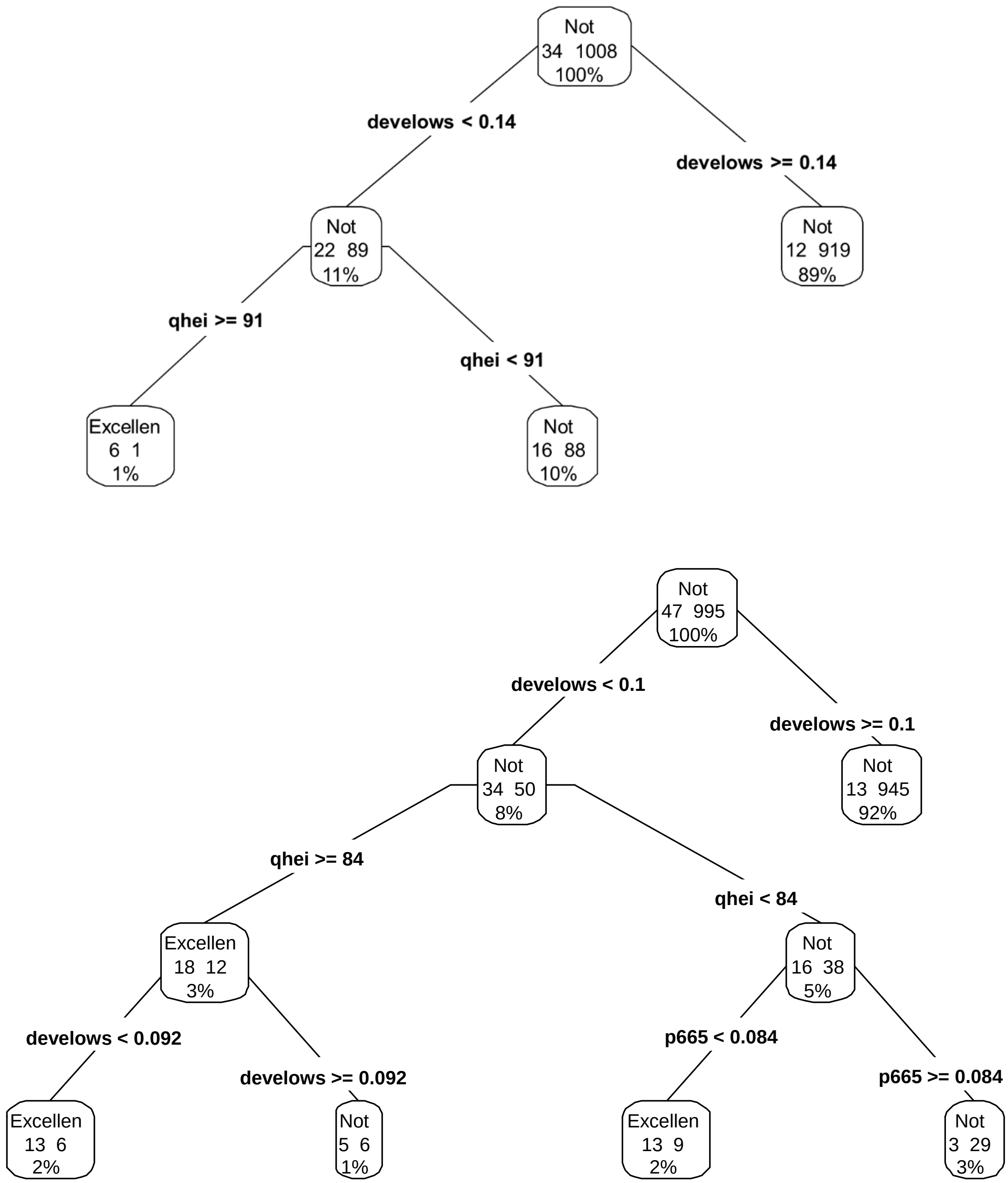



***Figure 48****. Results of a classification tree analysis with the mIBI Excellent threshold as the response variable (top) and fish IBI excellent narrative (bottom) and key stressor parameters from each stressor category as explanatory variables. Because a goal was the reduction of parameters, tree was limited to three levels.*

***Table 12**. FIT values based on the deviation between ambient stressor ranks vs. predicted stressor rank based on fish species or macroinvertebrate taxa for streams in the NE Illinois IPS study area. The algorithm for FIT calculation is summarized in the Chapter 2. The cell shading is related to FIT weighting coefficients: <0.1; <0.3; <1.0; <3.0; >3.0.*

| Stressor | FIT Value | Stressor | FIT Value |
|---|---|---|---|
| Impervious Land Use (500m) | 0.01 | Copper (Wat.) | 1.75 |
| QHEI Embeddedness Score | 0.03 | Lead (Wat.) | 2.11 |
| Urban Land Uses (WS) | 0.03 | Zinc (Sed.) | 2.22 |
| QHEI Overall Score | 0.04 | Benzo(g,h,i)perylene | 2.32 |
| QHEI Substrate Score | 0.04 | Indeno(1,2,3-cd)pyrene (Sed.) | 2.41 |
| QHEI Good Attributes | 0.04 | Copper (Sed.) | 2.42 |
| Total Phosphorus | 0.04 | Benzo(b)fluoranthene (Sed.) | 2.51 |
| Impervious Land Use (30m) | 0.04 | Turbidity | 2.61 |
| Impervious Land Use (30m Clipped) | 0.04 | Nickel (Sed.) | 2.67 |
| Conductivity | 0.05 | Manganese (Wat.) | 2.74 |
| QHEI Channel Score | 0.07 | Benzo(a)pyrene (Sed.) | 2.85 |
| QHEI Silt Cover Score | 0.07 | Pyrene (Sed.) | 2.85 |
| Developed Land Use (WS) | 0.07 | Voluble Suspended Solids | 2.81 |
| Minimum Dissolved Oxygen | 0.10 | Lead (Sed.) | 3.01 |
| Total Dissolved Solids | 0.10 | Nickel (Wat.) | 3.26 |
| Impervious Land Use (WS) | 0.10 | Benzo(a)anthracene (Sed.) | 3.48 |
| Hydro-QHEI Depth Score | 0.11 | Chrysene (Sed.) | 3.51 |
| QHEI Poor Habitat Attributes | 0.12 | Fluoranthene (Sed.) | 3.91 |
| Hydro-QHEI Overall Score | 0.13 | Strontium (Sed.) | 4.44 |
| Zinc (Wat.) | 0.13 | Dibenz(a,h)anthracene (Sed.) | 4.57 |
| Hydro-QHEI Current Score | 0.14 | Agricultural Land Use (WS) | 4.82 |
| TKN | 0.14 | Anthracene (Sed.) | 5.10 |
| QHEI Pool Score | 0.15 | Phenanthrene (Sed.) | 5.10 |
| Heavy Urban Land Use (WS) | 0.17 | Arsenic (Sed.) | 6.21 |
| Chloride | 0.17 | Chromium (Sed.) | 6.29 |
| QHEI Cover Score | 0.17 | Sulfate | 6.49 |
| BOD (5-Day) | 0.21 | Manganese (Sed.) | 7.08 |
| QHEI Riffle Score | 0.27 | Silver (Sed.) | 7.11 |
| Total Ammonia | 0.28 | Aluminum (Sed.) | 8.26 |
| Nitrate | 0.29 | Barium (Sed.) | 8.88 |
| Sodium | 0.29 | Arsenic (Wat.) | 9.19 |
| QHEI Gradient Score | 0.31 | Potassium (Wat.) | 10.13 |
| Total Suspended Solids | 0.32 | Cadmium (Sed.) | 11.0 |
| Maximum Dissolved Oxygen | 0.94 | | |
| Cadmium (Wat.) | 0.93 | | |
| Arsenic (Sed.) | 1.26 | | |

Very Poor are still considered to be predictive of Very Poor biological assemblages and are assigned that narrative in use attainment tables that now list the associated causes of impairment by narrative category. As the FIT value increases (i.e., >0.1 to 0.3) it signals increased variability (i.e., more outliers observed) the weighting factor declines to 0.8 and a stressor value of 9 (Very Poor) would be down weighted to a score of 7.2 (Poor) because the stress:response relationship had more outliers, presumably indicating a reduced ability to distinguish Poor vs. Very Poor assemblages, but still reflecting the cause of an impairment. A FIT value of >0.30-1.00 indicates a yet weaker SSD relationship and is assigned a lower weighting factor (X 0.6). This would change a stressor score of 9 (Very Poor) to a score of 5.4 (Fair). This approach guards against listing a cause of impairment as Very Poor where the uncertainty is higher, but still lists as a Poor or Fair cause depending on the FIT score. The goal for the IPS rankings is to focus first on causes where the certainty is higher. Parameters with FIT values of >3.00 are generally not used to assign causes of impairment. A summary of FIT values for 69 variables appears in Table 12. The Random Forest analyses added some valuable insights into key stressors, but local site, reach, and watershed perspectives and the spatial distribution of stressors (i.e., are they local, widespread, and contributing to cumulative impacts?) is still required for putting them into proper context.

Stressor relationships may become stronger as more data is added to the IPS databases hence the need for continued monitoring including reference sites. Some parameters may have a weak FIT score because the available data is incomplete across the full stressor gradient. For example, there are fewer data points at Excellent biological sites for parameters such as sediment PAHs and sediment metals. Because of the lack of sediment data for the Excellent narrative range only a Good narrative threshold was derived. There are other important variables (e.g., benthic chlorophyll a) where the current datasets are insufficient to develop an SSD ranking, again highlighting the need to continue to develop the regional dataset.

Caveats related to nutrient and organic enrichment parameters included a lack of data on sestonic and benthic chlorophyll a (measures of the primary biological response to elevated nutrients) and a reliance on grab sample D.O. data rather than continuous data that would better clarify the D.O. response signal. Developed land use data also provided strong FIT scores for certain parameters as follows; Impervious Land Use (500m) (FIT = 0.01) >Urban Land Uses (WS) (FIT = 0.03) >Impervious Land Use (30m Clipped) (FIT = 0.04) >Developed Land Use (WS) (FIT = 0.07). This contrasts with Agricultural Land Uses at the watershed scale where there was virtually no relationship (FIT = 4.82). There are no watersheds in the NE Illinois IPS study area that are predominantly forested and reference sites are typically located in areas of agricultural land uses with relatively low urban land and patchy forest cover.

The severity of the effect of some stressors (e.g., FIT Scores <0.1) could possibly mask the effects of other stressors. As more data is collected and as some of the more prevalent stressors are abated, the influence of masked stressors may become more apparent. As such, the FIT values and scores could change in future iterations of the IPS. More data will also improve the accuracy of assigning species and taxa as sensitive or tolerant to a particular stressor, which could also influence the FIT scores in future iterations of the IPS.

## Alternative Measures of Variable Importance: Random Forest Regression and Classification Tree Analyses

The preceding correlation, regression, and classification tree analyses provided additional insight into limiting variables, but the primary objective was to reduce the list of potential "responsible" stressors to the smaller number required for the Random Forest analyses.

### *Selection and Verification of Variables for the Restorability Rating Scores*

The FIT coefficients were used to derive the weighting factors for the Restorability, Susceptibility and Threat rankings and scores. There are varying degrees of autocorrelation between each of the stressor categories since they frequently overlap across NE Illinois. As an alternative approach for determining the relative importance of key stressors in explaining variation in fIBI and mIBI scores and General Use aquatic life attainment by extension, Random Forest (RF) classification was used to evaluate the relative importance of key stressors. Rather than the single regression and classification trees applied earlier in this Chapter, RF analyses randomly combine sites and variables over many trees that provide estimates of which variables occur as presumably being important and which variables contribute the most to the accuracy of the RF models. The RF analyses were also employed to identify which of the stressor variables from the prior battery of analyses remained as important stressors in the derivation of Restorability, Susceptibility, and Threat scores.

RF models are decision trees where a random component is employed via repeated tree building based on the bootstrap selection of different sites and the selection of different variables (split-variable randomization). Each time a split is calculated, the search for the most important split variable is limited to a random subset of all variables. The results from all trees are then averaged to identify variable importance to variation exhibited by the response variables. In essence this simulates what would have been done had a different subset of samples been used and a different mix of variables had been present. Most of the variables with FIT scores <1.00 were used in the RF analyses. Some of the sediment PAH, sediment metals, and water column metals parameters that had FIT scores >1.00 were also included to determine their importance in the RF analyses. The RF analyses included *regression* models with fIBI and mIBI scores, *classification* models with the narrative ranges of the fIBI and mIBI, and where full attainment of the General Aquatic Life Use benchmark occurred. Sites with only fish or macroinvertebrates (not both) were excluded from the General Use attainment classification models as attainment is based on meeting the biocriteria for both indices.

### Measures of Variable Importance

RF analyses (using the R statistics program RandomForest[13]) contain two measures of importance for regression and classification tree variables. The first assesses variables for the mean decrease in accuracy of a model when a variable is excluded and the mean decrease in

[13] https://www.stat.berkeley.edu/~breiman/RandomForests/

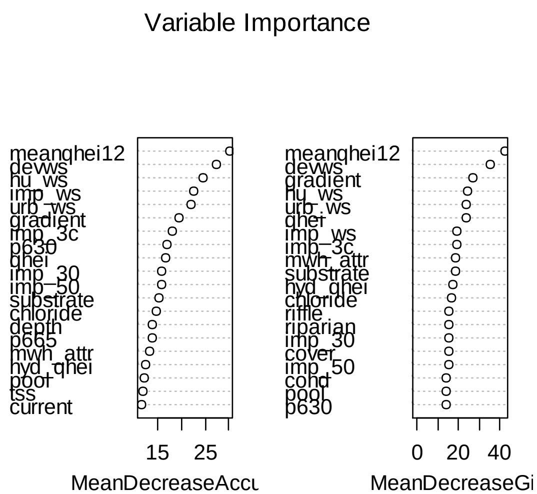
tree node impurity as a variable is excluded. This method uses error rate for classification (MSE for regression). The second method uses the node impurity as measured by the GINI index and for regression the residual sum of squares for classification. Variable accuracy is often considered the most important of these measures when determining variable importance. Node impurity focuses on whether a node is "purely" predicted by a variable or whether there are other branches below that node. As with other multivariate approaches, the relative ranking of variables is more important than the absolute values of the importance measures.

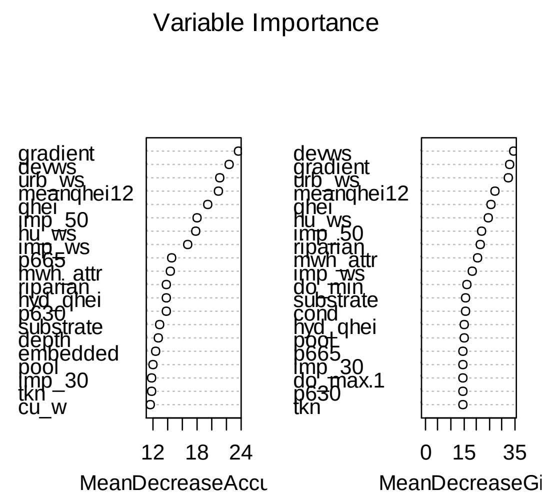
**Random Forest Model Results**

Figure 49 shows ordered plots of the accuracy and impurity importance measures for the fIBI (left) and mIBI (right) RF regression trees (based on 500 trees). Figure 50 illustrates a plot of variable importance for narrative fIBI and mIBI RF classification trees. Figure 51 illustrates plots of variable importance for a classification tree of attainment vs. non-attainment of the General Use for aquatic life. In addition to the stressor variables, two "non-stressor" variables, site drainage area (sq. mi.) and stream gradient (map gradient as ft./mi. as used in the QHEI) were used. Both of these naturally occurring factors can influence biological assemblages as well as on stressor strength or mode of effect. For example, small streams may have less dilution and be closer to or in more direct contact with potential sources of stress. Stream gradient can influence habitat features such as riffles and pools and how some pollutants exert their negative effects. The HUC12 scale mean QHEI score (mean QHEI12) was also included as an estimate of cumulative habitat quality.

Two variables that were not developed with the SSD analysis are at the upper end of importance in the RF regression trees for fIBI and mIBI – the mean HUC12 QHEI and drainage area (Figure 52). The HUC12 mean QHEI score is the top variable in the regression tree for fish and second for macroinvertebrates. Habitat not only exerts an effect at the local scale, but at the small watershed and reach scales reflecting a cumulative effect (Rankin 1995) and the RF results reveal that very phenomenon. Watersheds with extensively degraded habitat frequently have none or only a very few sensitive species or taxa that drive the positive metrics of the fIBI and mIBI. The reduction and outright elimination of viable populations of habitat sensitive species/taxa places an upper limit on biological condition resulting in lower index scores representative of the narrative ranges (Fair, Poor, Very Poor) below the General Use biocriteria.

The influence of drainage area on aquatic life condition is more complex. The IBI metrics of the fIBI are calibrated by stream size (e.g., species richness naturally increases with drainage area) so this is a reflection of a different phenomenon. Conversely the mIBI, which shows drainage area to be the most important variable in the RF analysis, is not calibrated by drainage area. Plots of the fIBI and mIBI vs. drainage area show threshold responses in both the fIBI (Figure 52, left) and the mIBI (Figure 52, right). Because the fIBI is calibrated by drainage area this suggests that the relationship is at least partially related to increased stressor effects in headwater streams, a pattern that could be tested with the addition of higher quality headwater stream reference sites in the future. Reference sites in the IPS study area averaged 290 sq. mi. and 69 of 72 reference samples were from sites >20 sq. mi. The smaller set of DRSCW reference

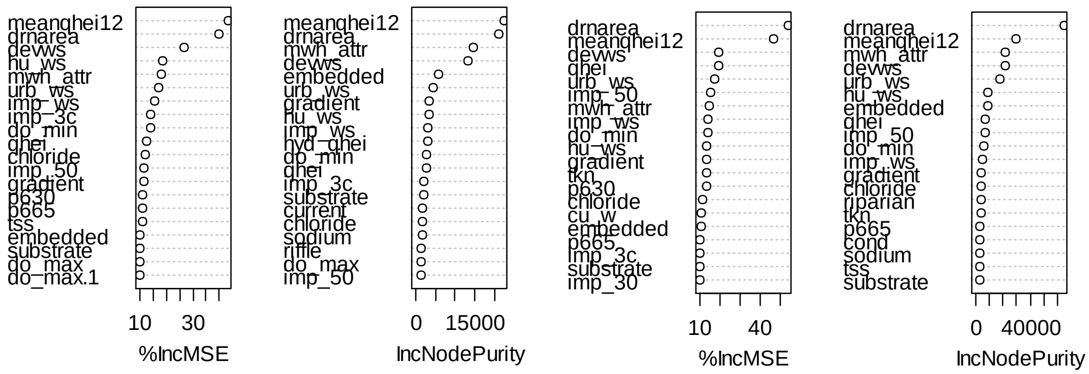



| Random Forest Classification Tree (fIBI Narrative Ranges) | | | | | | |
|---|---|---|---|---|---|---|
| Number of trees: 500 | | | No. of variables tried at each split: 6 | | | |
| OOB estimate of error rate: 30.04% | | | | | | |
| Confusion matrix: | | | | | | |
| fIBI Narrative | Excellent | Good | Fair | Poor | Very Poor | Class. Error |
| Excellent | 27 | 5 | 2 | 5 | 0 | 0.31 |
| Good | 9 | 7 | 14 | 4 | 0 | 0.79 |
| Fair | 9 | 6 | 67 | 38 | 0 | 0.44 |
| Poor | 1 | 1 | 22 | 378 | 30 | 0.13 |
| Very Poor | 0 | 0 | 2 | 89 | 73 | 0.55 |

| Random Forest Classification Tree (mIBI Narrative Ranges) | | | | | | |
|---|---|---|---|---|---|---|
| Number of trees: 500 | | | | No. of variables tried at each split: 6 | | |
| OOB estimate of error rate: 39.04% | | | | | | |
| Confusion matrix: | | | | | | |
| fIBI Narrative | Excellent | Good | Fair | Poor | Very Poor | Class. Error |
| Excellent | 1 | 23 | 0 | 0 | 0 | 0.96 |
| Good | 7 | 173 | 78 | 71 | 0 | 0.27 |
| Fair | 0 | 54 | 53 | 4 | 0 | 0.62 |
| Poor | 0 | 6 | 44 | 220 | 8 | 0.21 |
| Very Poor | 0 | 0 | 5 | 33 | 9 | 0.81 |

***Figure 49**. Variable importance plots of the increase in mean square error or accuracy (left) and increase in node purity (right) for random forest regression tree analyses for the fIBI (left plots) and mIBI (right plots) for wadeable streams in the NE Illinois IPS study area. Average percent variance of that the models explained located at bottom.*




| Type of random forest: Regression | |
|---|---|
| Fish Assemblage: **fIBI** | Macro. Assemblage: **mIBI** |
| Number of trees: 500 | Number of trees: 500 |
| No. of variables tried at each split: 13 | No. of variables tried at each split: 13 |
| Mean of squared residuals : 33.34814<br>% Var explained: **72.99** | Mean of squared residuals: 86.329<br>% Var explained: **68.96** |

***Figure 50****. Variable importance plots of mean decrease in accuracy (left plots) and decrease in purity (GINI) score (right plots) for random forest classification tree analysis for narrative ranges (excellent, good, fair, poor, and very poor) for fIBI (left) and mIBI (right)for wadeable streams in the NE Illinois IPS study area. The average OOB error rate and confusion matrix of narrative range of the indices are located at the bottom.*

sites average 72.9 sq. mi. with only one site <20 sq. mi. Smaller streams have less dilution and are more easily inundated by riparian encroachment, instream modifications, and altered hydrology than are larger streams. They are more directly influenced by adjacent land uses that deliver pollutants such as chlorides, nutrients, suspended materials, PAHs, and metals. A cursory examination of the number of pollutants by stream size showed variable results, but there were higher concentrations of selected pollutants at headwater sites for parameters such as TP and chlorides and with lower QHEI scores. The strong influence of drainage area was documented in statistical outputs of the earlier and more spatially restricted DRSCW IPS framework (Miltner et al. 2010). Another non-stressor variable that had some importance in the RF analyses was gradient (as used in the QHEI). For both fish and macroinvertebrates lower IBI scores were evident at the lowest gradient sites which more frequently had excessive sediment deposition and low D.O. Higher gradients occurred at sites draining <30 sq. mi. which is likely a surrogate for the same effects described earlier for headwater streams.

Table 13 reports the rankings of the top 20 variables for measures of goodness of FIT (values <0.32) and Random Forest (RF) importance ranks. The mean HUC12 QHEI was at or near the top of each RF analysis illustrating the importance of reach and small watershed level cumulative habitat. After stream size and HUC12 QHEI, the urban related developed and impervious land use variables at both the watershed and 500m spatial buffer scales were important for both the fIBI and mIBI. This was followed by the site QHEI score and embeddedness score which is similar to the FIT scores from the univariate analyses in Chapter 3. While the exact rank order of the importance measures between the FIT scores and the RF regression scores is not identical (Table 12 vs. Figure 49 and as summarized in Table 13) the pattern suggests that multiple stressors nearly always contribute to variation in the fIBI and mIBI, habitat attributes (substrate and embeddedness) in particular, along with chlorides, D.O., and nutrients. A similar pattern existed where the RF classification tree analyses focused on narrative ranges of the fIBI and mIBI. The RF classification tree analysis using sites attaining the General Use (Good) for aquatic life as the response variable, the stressor importance ranks were similar, but included more land use variables (Figure 52; Table 13). The appearance of stressors from the major categories including habitat, ionic strength, nutrients, and organic enrichment suggests that in the various combinations of sites used in the RF analysis each of these categories contributed to aquatic life impairment and condition at certain sites. This result suggests that local stressor analyses at the HUC12 watershed, reach, and site scale are essential to understanding the limiting effects of stressors in the NE Illinois IPS study area. The predominance of land use as an important variable along with mean HUC12 QHEI points to the importance of cumulative stressor effects in these watersheds. The apparent "nestedness" of impairments and stressors underscores the importance of measuring and understanding watershed scale and cumulative impacts. One form conclusion is readily apparent – watershed scale habitat as represented by the mean HUC12 QHEI is a critical and pervasive factor, thus any watershed restoration project will need to take this into account if success is to be expected.

These results also point out that monitoring designs that do not measure stressors and responses at the spatial scales used for the IPS will likely reveal different stressors, perhaps to the point that the most important stressors will be overlooked. The IPS Dashboard (MBI 2022b) is designed to guide a user through the data at the HUC12 watershed scale and then enable "drilling down" to the reach and site-specific scales for data about attainment status, biological assemblages, stressors, and measures of Restorability, Susceptibility, and Threat.

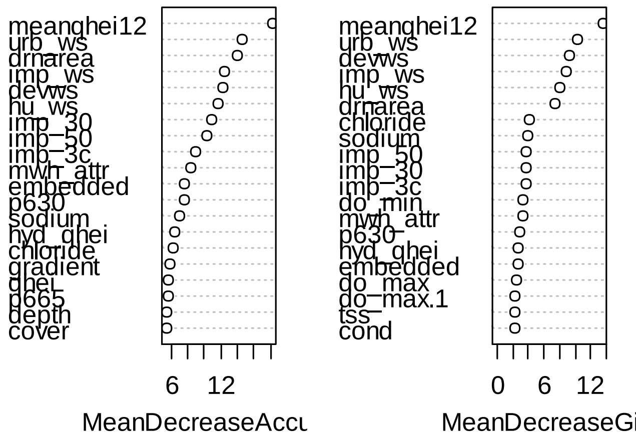
***Table 13**. Measures of Goodness of FIT (values <0.32) and Random Forest (RF) importance ranks (1-20) for key Illinois IPS stressors. The top 5 ranked forest variables in each analysis are in blue boldface type.*

| | | Regress. & Class. Tree | | RF Regression Tree Importance Rank MSE[14]/Impurity | | RF Classification Tree Importance Rank MSE[14]/Impurity | | |
|---|---|---|---|---|---|---|---|---|
| **Stressor** | **FIT Score** | **Fish** | **Macros** | **fIBI** | **mIBI** | **Fish by Narrative** | **Macros. by Narrative** | **General Use Attain-ment** |
| Huc12 Mean QHEI | - | - | - | **1/1** | **2/2** | **1/1** | **3/3** | **1/1** |
| Impervious Land Use (500m) | 0.01 | ✓ | ✓ | 12/20 | 6/9 | 11/17 | 6/7 | 8/9 |
| QHEI Embeddedness Score | 0.03 | ✓ | ✓ | 17/**5** | 16/7 | - | 16/ - | 11/16 |
| Urban Land Uses (WS) | 0.03 | | | 6/6 | **5/5** | **5/5** | **3/3** | **2/2** |
| QHEI Overall Score | 0.04 | ✓ | ✓ | 10/12 | **4**/8 | 9/6 | **5/5** | 17/ - |
| QHEI Substrate Score | 0.04 | ✓ | ✓ | 17/14 | 19/20 | 12/10 | 14/12 | - |
| QHEI Good Attributes | 0.04 | ✓ | ✓ | - | - | - | - | - |
| Total Phosphorus | 0.04 | ✓ | ✓ | - | 17/15 | 15/ - | 9/16 | 18/ - |
| Impervious Land Use (30m) | 0.04 | - | - | - | 20/ - | 10/15 | 18/ - | 7/11 |
| Impervious Land Use (30m Clipped) | 0.04 | - | - | 8/13 | 17/ - | 7/8 | - | 9/10 |
| Conductivity | 0.05 | ✓ | ✓ | - | - | - /18 | - /13 | - /20 |
| QHEI Channel Score | 0.07 | ✓ | ✓ | - | - | - | - | - |
| QHEI Silt Cover Score | 0.07 | | | - | - | - /16 | - | - |
| Developed Land Use (WS) | 0.07 | ✓ | ✓ | **3/4** | **3/4** | **2/2** | **2/1** | **5/3** |
| Minimum Dissolved Oxygen | 0.10 | | | 9/11 | 9/10 | - | - | - /12 |
| Total Dissolved Solids | 0.10 | | | - | - | - | - | - |
| Impervious Land Use (WS) | 0.10 | | | 7/9 | 8/11 | **4**/7 | 8/10 | **4/4** |
| Hydro-QHEI Depth Score | 0.11 | | | - | - | 14/ - | 15/ - | 19/ - |
| QHEI Poor Habitat Attributes | 0.12 | ✓ | ✓ | **5/3** | 7/**3** | 16/9 | 10/9 | 10/12 |

[14] MSE = Mean Square Error which is average of the summation of the squared difference between the actual output value and the predicted output value.

***Table 13**. Measures of Goodness of FIT (values <0.32) and Random Forest (RF) importance ranks (1-20) for key Illinois IPS stressors. The top 5 ranked forest variables in each analysis are in blue boldface type.*

| | | Regress. & Class. Tree | | RF Regression Tree Importance Rank MSE[14]/Impurity | | RF Classification Tree Importance Rank MSE[14]/Impurity | | |
|---|---|---|---|---|---|---|---|---|
| **Stressor** | **FIT Score** | **Fish** | **Macros** | **fIBI** | **mIBI** | **Fish by Narrative** | **Macros. by Narrative** | **General Use Attain-ment** |
| Hydro-QHEI Overall Score | 0.13 | | | - /10 | - | 17/11 | 11/14 | 14/15 |
| Zinc (Wat.) | 0.13 | ✓ | ✓ | - | - | - | - | - |
| Hydro-QHEI Current Score | 0.14 | | | - /15 | - | 20/ - | - | - |
| TKN | 0.14 | ✓ | ✓ | - | 12/15 | - | 19/20 | - |
| QHEI Pool Score | 0.15 | | | - | - | 18/19 | 17/15 | - |
| Heavy Urban Land Use (WS) | 0.17 | | | 4/6 | 10/6 | 3/4 | 7/6 | 6/5 |
| Chloride | 0.17 | ✓ | ✓ | 11/16 | 14/13 | 13/12 | - | 15/7 |
| QHEI Cover Score | 0.17 | | | - | - | - /16 | - | 20/ - |
| BOD (5-Day) | 0.21 | | | - | - | - | - | - |
| QHEI Riffle Score | 0.27 | | | - /18 | - | - /13 | - | - |
| Total Ammonia | 0.28 | ✓ | ✓ | - | - | - | - | - |
| Nitrate | 0.29 | ✓ | ✓ | 14/ - | 13/ - | 8/20 | 13/19 | 12/14 |
| Sodium | 0.29 | | | - /17 | - /18 | - | - | 13/8 |
| QHEI Gradient Score | 0.31 | | | 13/7 | 11/12 | 6/3 | 1/2 | 16/ - |
| Total Suspended Solids | 0.32 | | | 16/ - | - /19 | 19/ - | - | - /19 |




| Random Forest Classification Tree | | | |
|---|---|---|---|
| Number of trees: 500 | | No. of variables tried at each split: 6 | |
| OOB estimate of error rate: 4.44% | | | |
| Confusion matrix: | | | |
| Attainment Status | Non | Attains | Class. Error |
| Non | 704 | 14 | 0.019 |
| Attains | 21 | 50 | 0.296 |

***Figure 51**. Variable importance plots of mean decrease in accuracy (left) and decrease in purity (GINI) score (right) for random forest classification tree analysis for the attainment of the Illinois General Aquatic Life use for wadeable streams in the NE Illinois IPS study area.*

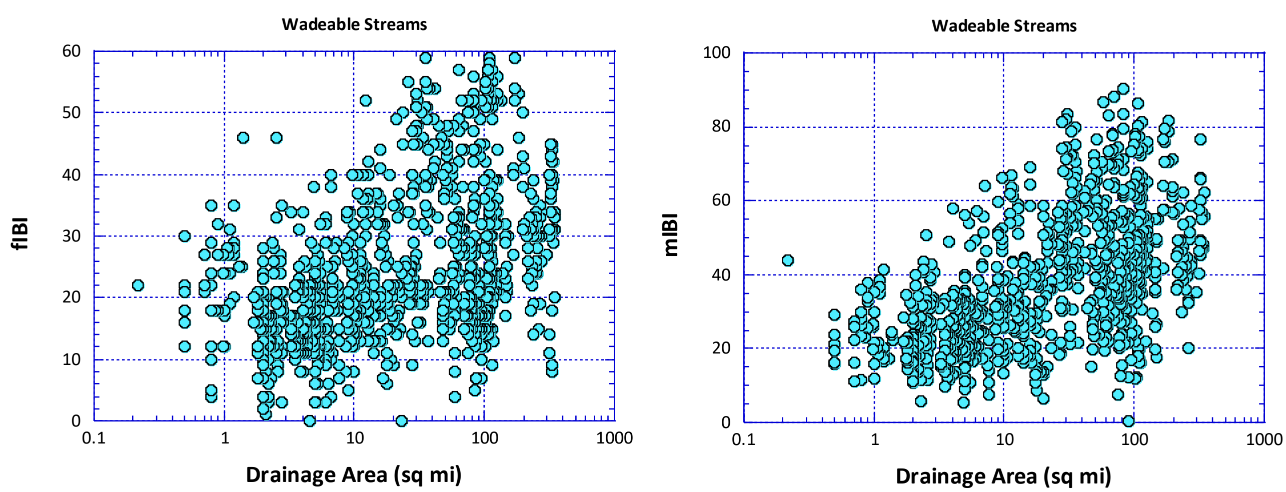



***Figure 52**. Plots of drainage area (sq. mi) vs. fIBI (left) and mIBI (right) for wadeable sites in the NE Illinois IPS study area.*

# Chapter 5. Restorability and its Association with Biological Condition

## INTRODUCTION

The goal of deriving measures of Restorability, Susceptibility, and Threat is to link factors back to the biological benchmarks that correspond to the relative tractability of remediating impairments (Restorability) or the need and urgency to protect attaining sites against future degradation (Susceptibility and Threat). This Chapter examines the association between the Restorability ranking score and subcomponents of the fIBI and mIBI and a discussion of the potential applicability of the Restorability, Susceptibility and Threat factors in NE Illinois watersheds.

### Data Confidence Measure

Confidence in Restorability scores is related to the quality and amount of data available for assessing condition and stressor identification. Data confidence for the IPS is derived for the biological data and each category of stressor data. The data in the IPS is built around sites that have sufficient biological data to determine attainment status that is paired with sufficient data about chemical and physical stressors. At the site-specific scale, sites with both fish and macroinvertebrate receive the highest rank of 5 which imparts high confidence. Sites with either fish or macroinvertebrate data receive a score of 3, moderate confidence. Each stressor category individually receives a score of 1 (low confidence) to 5+ (high confidence) depending on the number of stressor variables in a category. The stressors with the best FIT scores for a category accrue more points and higher confidence scores indicating adequate coverage of key stressor variables.

### Weighting Restorability Subcomponents

The ***Restorability Ranking Scores*** are based on a composite of individual stressor scores that can be divided into three main categories: biological variables, chemical variables, and physical variables (habitat and land use). In this section each of the individual components are defined and described along with the rationale for weighting the variables. Weightings are not static and trial applications of these factors in real world applications could result in changes or even alternate forms of the rankings.

### Biological Components

#### *HUC12 Biological Factors*

Based on analyses from Ohio, Indiana, Illinois and Minnesota, there is a strong cumulative effect of stressors (*e.g.,* habitat) that are associated with biological performance at watershed scales. Sites are more likely to have good-excellent biological index scores in watersheds where other sites also perform well and stressors are low. Conversely, where sites have poor biological performance, neighboring sites in a watershed typically also have poor or very poor biological performance. Watersheds with high cumulative stressor effects such as widespread habitat loss limit the reproduction and survival of sensitive species/taxa resulting in lower biological index scores. Cumulative biological variables and scoring criteria are listed in Table 14.

***Table 14**. Cumulative fIBI and mIBI Restorability score weighting criteria.*

| Variables | Scoring (0.1-10; Best->Worst) | Score |
|---|---|---|
| Cumulative Percentage of fIBI and mIBI Scores Attaining General Use Biocriteria | ≥ 80% Attaining; Mean Attaining (fIBI ≥ 50; mIBI ≥ 73) | 1.0 |
| | ≥ 50 - <80% Attaining; Mean Attaining (fIBI ≥ 50; mIBI ≥ 73) | 1.5 |
| | ≥ 25 - <50% Attaining; Mean Attaining (fIBI ≥ 50; mIBI ≥ 73) | 2.0 |
| | <25% Attaining; Mean Attaining (fIBI ≥ 50; mIBI ≥ 73) | 2.5 |
| | ≥ 50 % Attaining; Mean Attaining (fIBI ≥ 41-50; mIBI ≥ 41.8-73) | 3.0 |
| | ≥ 25 - <50% Attaining; Mean Attaining (fIBI ≥ 41-50; mIBI ≥ 41.8-73) | 3.5 |
| | < 25 % Attaining; Mean Attaining (fIBI ≥ 41-50; mIBI ≥ 41.8-73) | 4.0 |
| | 0 % Attaining; Mean Attaining (fIBI ≥ 30-<41; mIBI ≥ 30 - <41.8) | 5.0 |
| | 0 % Attaining; Mean Attaining (fIBI ≥ 20-<30; mIBI ≥ 20 - <30) | 6.0 |
| | 0 % Attaining; Mean Attaining (fIBI ≥ 10-<20; mIBI ≥ 10 - <30) | 8.0 |
| | 0 % Attaining; Mean Attaining (fIBI <10; mIBI < 10) | 10.0 |
| No Biological Data | | 4.0 |

***Local Biological Factors***

The restorability ranking also considers local biological condition as a factor in the ranking and using the 0.1 (best) to 10 (worst) ranking of the fIBI and mIBI independently. Here a ranking of 2 is associated with an exceptional threshold and a ranking of 4 is associated with the General Use (Good) biocriteria for each index with a linear interpolation of scores down to the minimum score which is 10. [Total Points: 20, 10 for fIBI, 10 for mIBI]

## Chemical Stressors

Chemical stressors are a component of the Restorability rankings and contribute based on the most severe parameter that contributes to a chemical category which also factors in the FIT rankings. For sites without chemical data, the ranking is assumed to be 3 for a parameter category which is in the middle range of the ranking of the General Use (>2-4). Sites with better than the General Use thresholds can contribute a score <4 which reflects lower chemical stress. These are linearly interpolated between the minimum value and the threshold values of 2 and 4, or from 4 where the stressor gradient does not include the Excellent range.

## Physical/Land Use Factors

***Physical Habitat***

In Midwest states physical habitat is often the most limiting stressor to aquatic life in streams and rivers. Local habitat condition is measured directly using the QHEI and the ranking (0.1 (best) to 10 (worst) for the overall QHEI score is a subcomponent of the Restorability, Susceptibility and Threat ranking scores. Although the QHEI is the most frequent limiting habitat parameter, occasionally QHEI metrics such as channel condition or QHEI substrate score may be more limiting. If so, they are used in place of the QHEI ranking score. In addition each metric score and ranking is available in the Power BI dashboard page for habitat stressors and these are useful for interpreting the nature of the habitat impact at a site, reach, or watershed scales.

***Cumulative Habitat Impacts***

Previous analyses in Midwest States (e.g., Ohio, Minnesota, and Indiana) have shown cumulative effects of habitat loss on biological condition. The average QHEI score within a HUC12 watershed is such an example. The difference between the cumulative biological restoration ranking scores and the QHEI cumulative rankings is that the biological measures in a HUC12 watershed in the year in which the biological data was collected is used because of known trends in the data in recent times. Cumulative habitat measures use all available habitat data which provides a more robust indicator of habitat conditions, provided there are sufficient sites sampled. Because of its importance it is double weighted (max. of 20 points). This is supported by the RF analyses where HUC12-scale mean QHEI habitat (along with drainage area) were consistently among the most important variables, emphasizing the importance of having indicators of cumulative impacts. This is an important concept because it tempers expectations of short-term biological recovery from locally focused restoration efforts in watersheds where cumulative impacts are dominant. Smith et al. (2016) recognized that cumulative impacts can limit ecological condition and recommend a concept that melds ecological restoration with "benefits from incorporating societal outcomes into urban stream restoration projects." Smith et al. (2016) propose "urban stream renovation as a flexible stream improvement framework in which short-term ecological and societal outcomes are leveraged to achieve long-term ecological objectives." A key aspect of this approach is that long-term cumulative ecological outcomes are not abandoned, but recognizes that short-term recovery in dense urban areas is unlikely without the support of local stakeholders and the public for sustaining long-term efforts to restore streams, riparian areas, and floodplains. The consideration of developing subcategories of the General Use for aquatic life would provide a better regulatory structure to the approach of Smith et al. (2016) which compared their approach to the more typical one-size-fits-all "reference site" approach.

## Land Use Data

Multiple studies have shown the strong correlations between certain land use variables and biological data to the extent that the widespread aquatic life impairment in urban watersheds has been coined the "urban stream syndrome" (Walsh et al. 2005). As was described earlier and underscored by the RF statistical analyses they are an influential component of the NE Illinois IPS. The most limiting of the watershed and spatial buffer measures of land use are weighted as components of the Restorability ranking score and most factors receiving full weighting (Table 4).

## Scaling of Restorability Ranking Scores

The raw Restorability ranking scores were rescaled from 0 to 100 with 100 being the "most" restorable and zero the least. The rescoring was done based on the highest and lowest restorability rating score at sites that were impaired during the initial analysis of the data. Restorability rankings only apply to impaired waters, however scaled scores above 100 can be calculated at sites that fully attain the General Use and these are used in the plots of the Restorability ranking score vs. individual biological stressor parameters and provide a greater range of values in these plots.

The Susceptibility ranking only applies to waters that fully attain the biological thresholds for the General Use thresholds for both the fIBI and mIBI. With Susceptibility the most biologically diverse and least impacted waters are considered the most susceptible to stressors because such streams are uncommon in the study area and often associated with lowest stressor "load." A companion the Susceptibility ranking is the Threat ranking which also applies only to attaining waters. A site is considered more susceptible where biological index scores are high and stressor levels are low. Threat rankings are nearly the converse where Threat scores are higher where stressor scores are elevated. The highest threat is where there are more stressors and/or stressors are at levels (Poor, Very Poor) associated with biological impairments.

## Restorability Ranking Scores and Biological Indicators

The basis of the Restorability ranking scores lies in the relationships between species-specific stressor metrics and stressors linked to the sensitive species metrics expected at sites meeting the Illinois General Use aquatic life benchmarks and an Excellent benchmark derived for the upper tier of aquatic assemblage condition. The fIBI and the mIBI are multimetric indices comprised of metrics that reflect the aggregate of assemblage responses to stressors and disturbance. Because the IPS study area has a wide array and variety of human impacts it can be useful to examine how selected fIBI and mIBI metrics respond to the Restorability Ranking Scores derived by this study.

In headwater and wadeable streams, both the fIBI and mIBI show a positive relationship with the Restorability Ranking Score where the ranking is scaled to 0-100 with zero being the least restorable and 100 the most restorable scenario (Figures 53-57). In these graphs points are coded at non-attaining sites by the most limiting stressor at the site scale (highest 0.1-10 stressor rating). Land use variables were excluded except in one figure to demonstrate its dominance (Figure 56, bottom right). Not surprisingly, the most restorable sites have better existing biological quality, better habitat conditions, and fewer chemical threshold exceedances than those of the lower quality ranges (i.e., fair, poor, or very poor levels). Sites with lower restorability are associated with poorer existing biological conditions, are more likely to have degraded habitat, more frequent and higher magnitude chemical threshold exceedances, and are more likely to be in urban developed watersheds and with and higher IC. In these figures the primary correlative causal factor was the most severe of the all the weighted stressor ranks

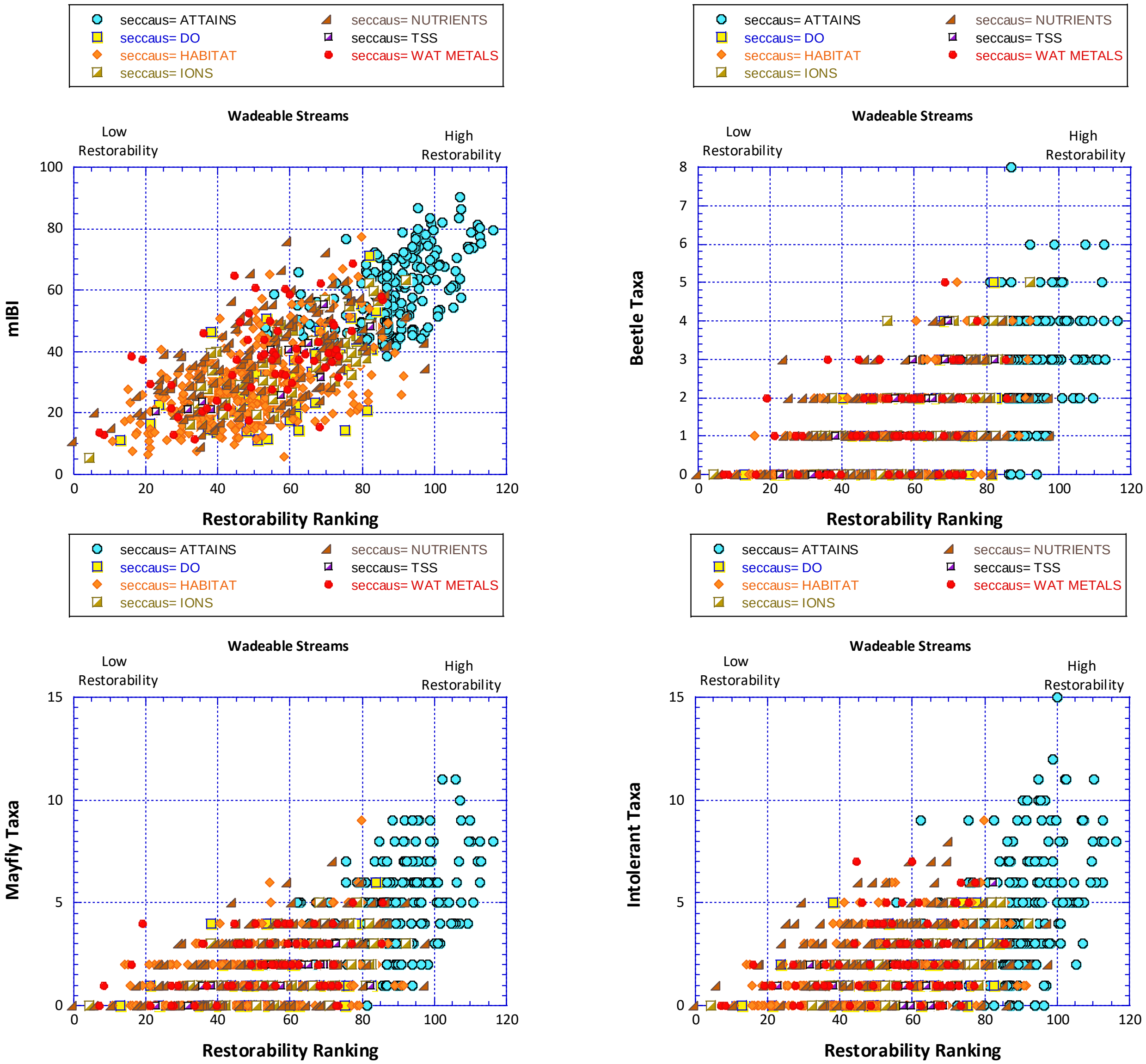



***Figure 53****. Plots of the IPS Restorability Ranking vs. the mIBI (top, left), beetle taxa (top, right), mayfly taxa (bottom, left) and intolerant taxa (bottom, right). Point are coded as attaining the Illinois General Use (blue) or if not attaining by the stressor category with the most severe ranking score excluding land use parameters.*

(>4-10). We ran this analysis without land use data since the land use factors are more of a "source" of stressors (e.g., via runoff) although they also serve as a surrogate for an altered flow regime. Nonetheless, the key stressor parameter plots without land use reveal multiple proximate stressors limiting to aquatic life. The prevalence of multiple stressors in the database is reflected in the fact that key variables were scattered throughout the breadth of the relationships and did not cluster at either end of the Restorability gradient her. Data in the IPS Dashboard provides results of all elevated stressors for each site in the NE Illinois IPS.

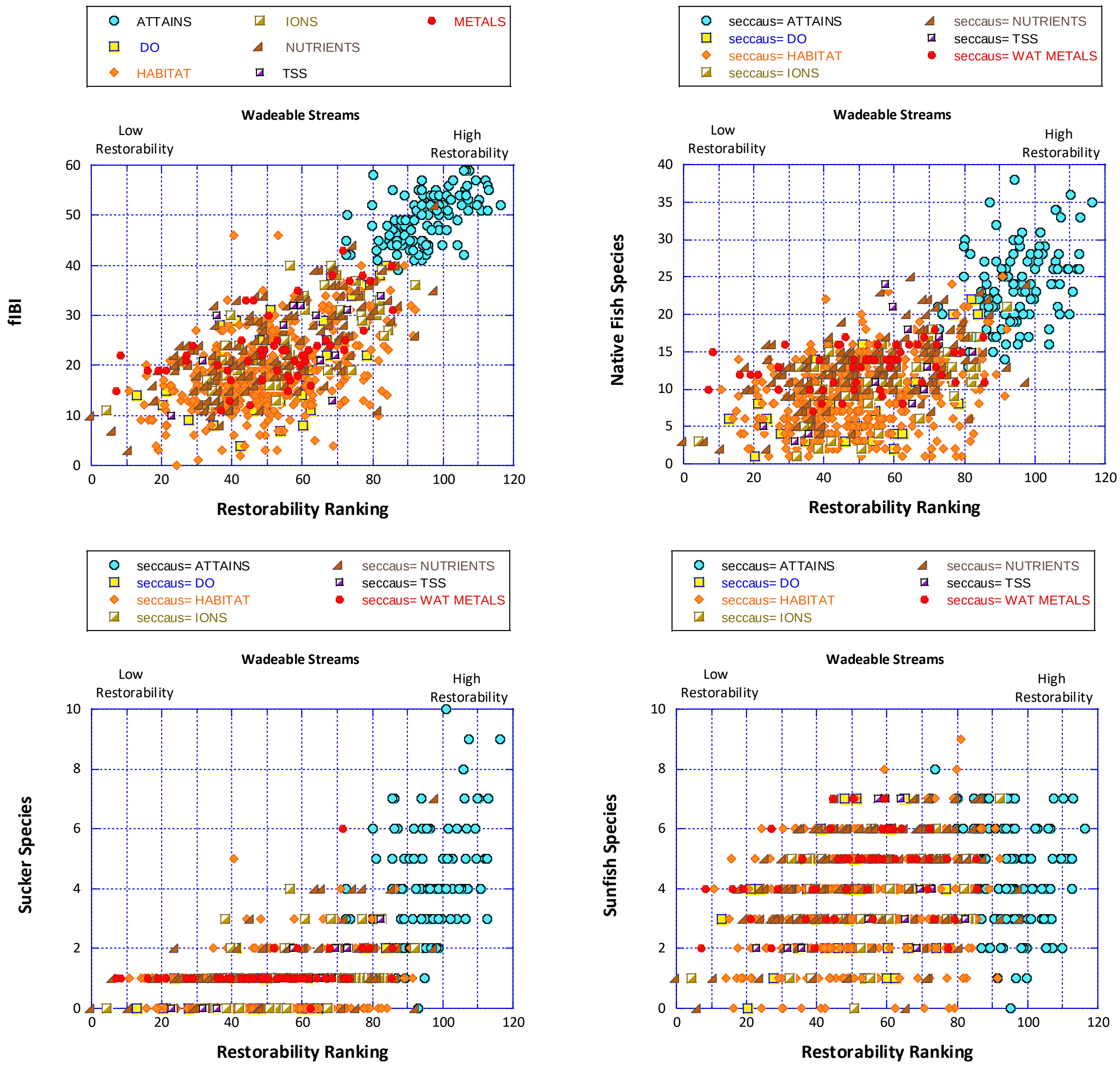



***Figure 54**. Plots of the IPS Restorability Ranking vs. the MIB index (top, left), % scrapers (top, right), % EPT individuals (bottom, left) and all (total) taxa (bottom, right). Point are coded as attaining the Illinois General Use (blue) or if not attaining by the stressor category with the most severe ranking score excluding land use parameters.*

## Low Restorability Ranking Score Sites

Table 15 lists the 15 “least restorable” sites by year on the basis of the Restorability ranking scores. The complete list of the Restorability ranking scores and other supporting information is available on the Restorability page of the NE Illinois IPS Dashboard. Eight (8) of the 15 sites are in Addison Creek, two are in the Arlington Branch of Salt Creek, and two on the East

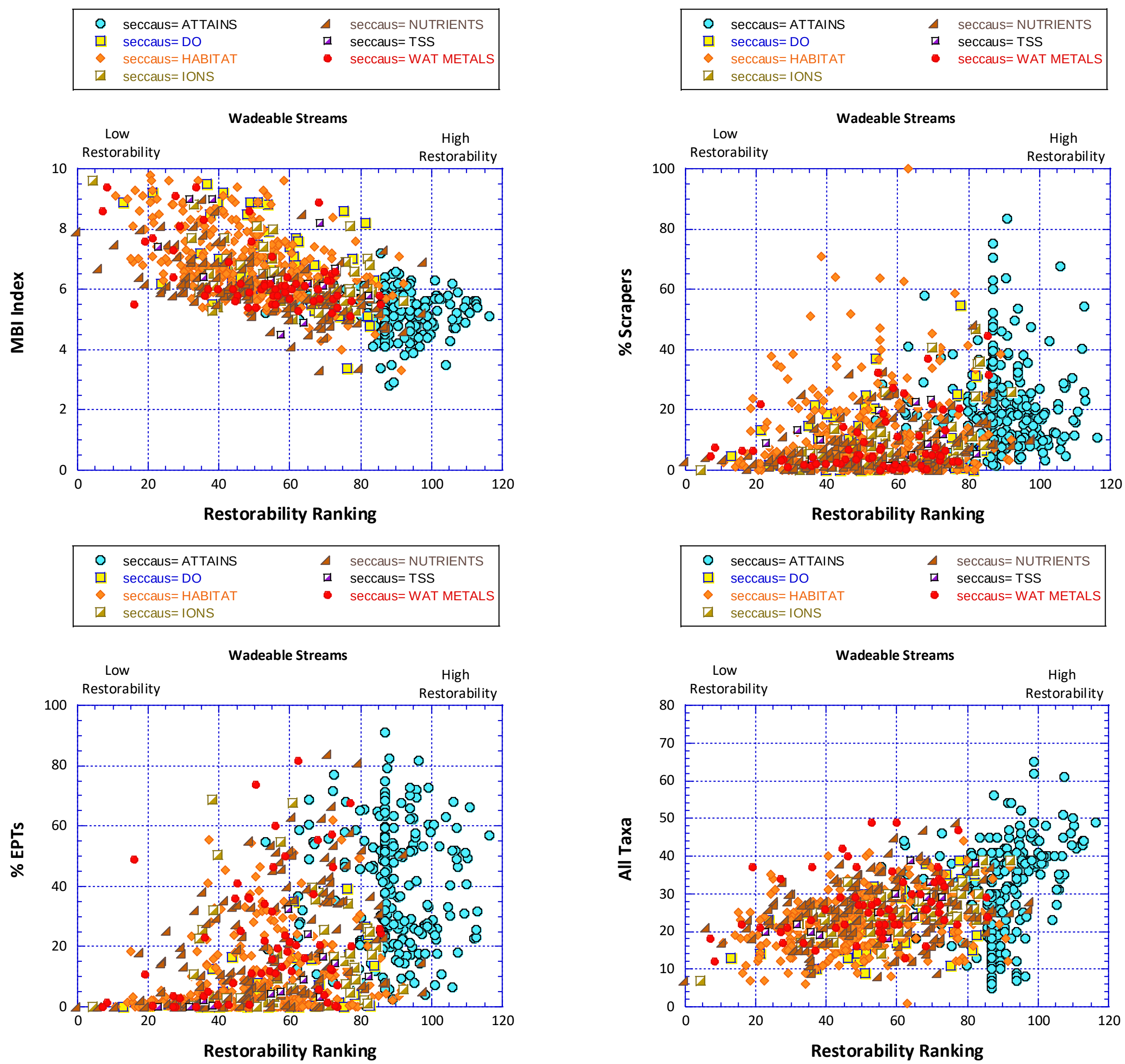



*Figure 55. Plots of the IPS Restorability Ranking vs. the fIBI (top, left), native fish species (top, right), sucker species (bottom, left) and sunfish species (bottom, right) in streams of less than 350 sq. mi. drainage. Points are coded as attaining the Illinois General Use biocriteria (blue) or if not attaining, by the stressor category with the most severe ranking score excluding land use parameters.*

Branch of Salt Creek. These sites were similar to those in the original IPS (Miltner et al. 2010). All of these sites had Poor or Very Poor biological index scores and multiple stressors ranked as Poor or Very Poor. Each site also had one or more land use stressors (impervious cover, urban, heavy urban, or developed) in the Very Poor range. All had Poor or Very Poor habitat attributes (e.g., substrate, channel metrics). Ten (10) of the 15 sites had Poor or Very Poor low D.O. or $BOD_5$ stressors. These sites also had Fair, Poor, or Very Poor ionic strength threshold exceedances (chloride, conductivity). Seven (7) of the sites had Fair, Poor, or Very Poor water

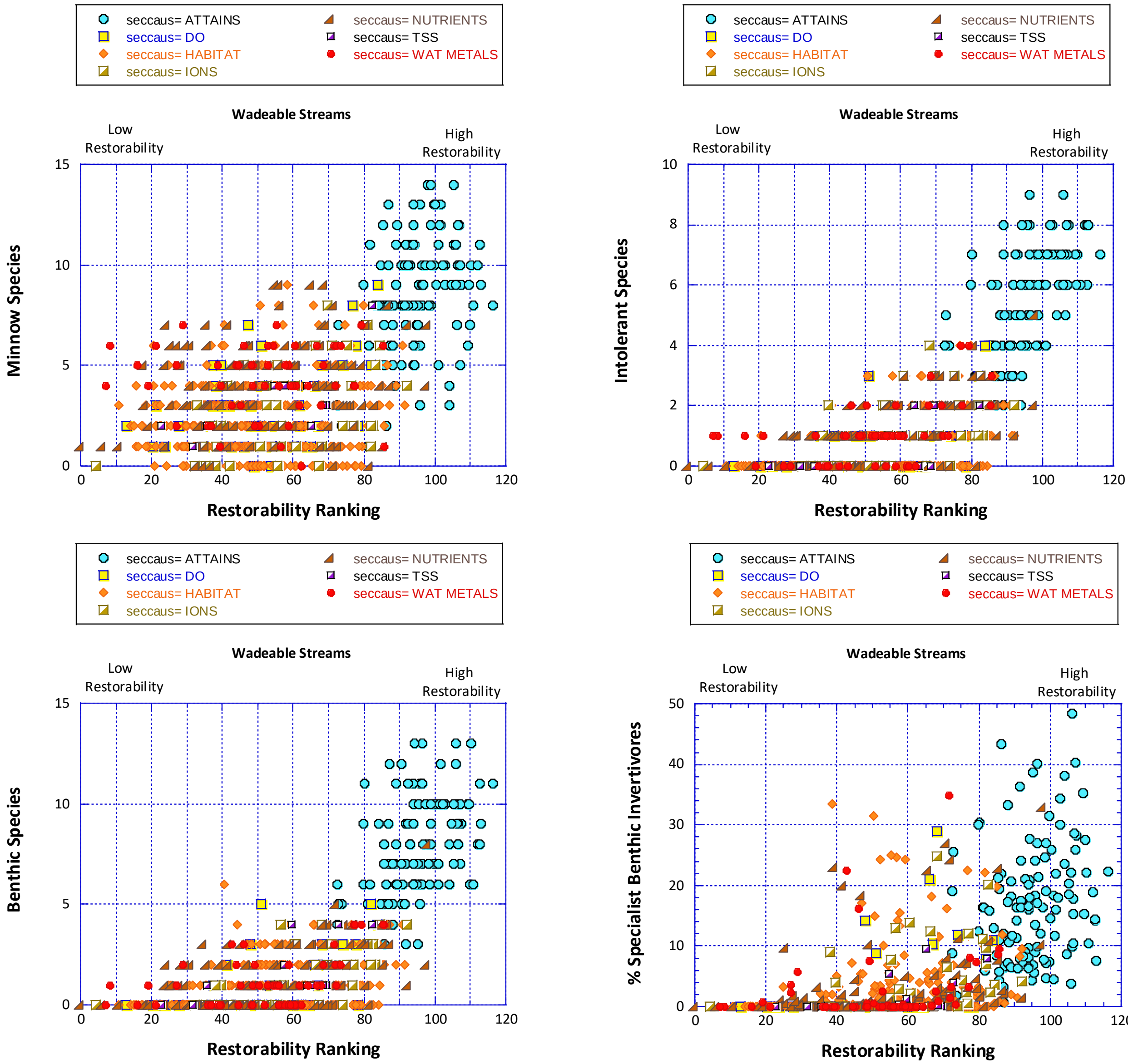



***Figure 56.*** *Plots of the IPS Restorability Ranking vs. the number of minnow species (top, left), intolerant fish species (top, right), benthic species (bottom, left) and % specialized benthic invertivores (bottom, right) in streams of less than 350 sq. mi drainage. Point are coded as attaining the Illinois General Use biocriteria (blue) or if not attaining by the stressor category with the most severe ranking score excluding land use parameters.*

column or sediment metal concentrations. Ten (10) of the 15 sites also had Fair, Poor, or Very Poor levels of organic enrichment and nutrients as measured by TKN or TP. Only three (3) of the 15 sites had Fair, Poor, or Very Poor levels of TSS or turbidity. Thus these sites are characterized by the strong impervious land use signature, multiple key stressors are elevated, and all likely contribute to aquatic life impairment. As an aside, TSS and turbidity, which are often used as a

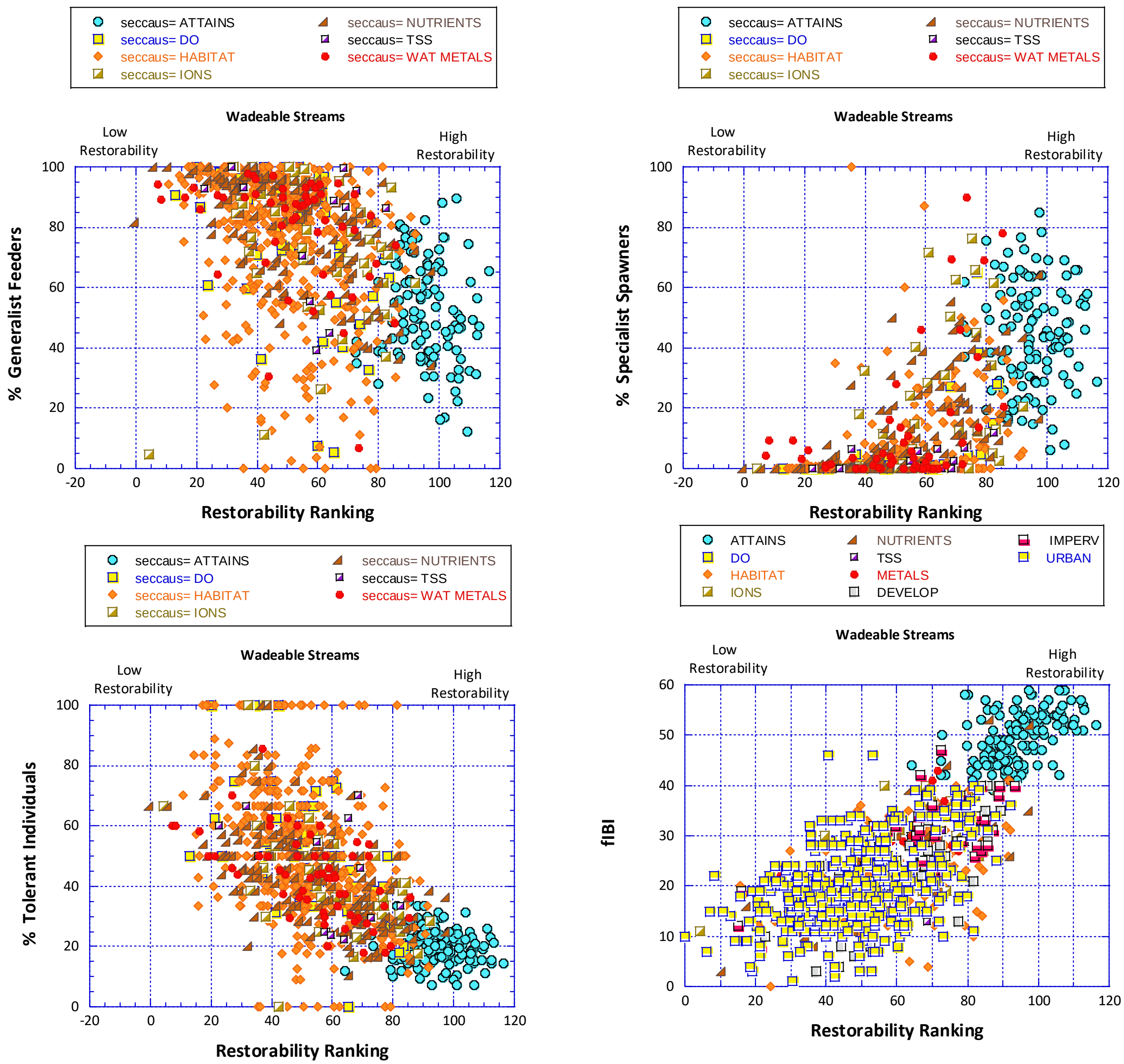



***Figure 57**. Plots of the IPS Restorability Ranking vs. the % generalist feeders (top, left), % specialist spawners (top, right), % tolerant individuals (bottom, left) and fIBI show land use (bottom, right) in streams of less than 350 sq. mi drainage. Point are coded as attaining the Illinois General Use biocriteria (blue) or if not attaining by the stressor category with the most severe ranking score excluding land use parameters.*

sole surrogate for identifying, prioritizing, and modeling stormwater impacts, were less important in identifying the most degraded and least restorable sites in the study area.

## High Restorability Ranking Score Sites

Table 15 list the top 15 "most restorable" sites by year on the on the basis of the Restorability Rating Score. These sites are all partially impaired (i.e., one of the two indices meets its benchmark) or only one assemblage index was available at a site and it was impaired, but close

***Table 15**. IPS sites with the lowest Restorability ranking scores, identified stressors by stressor magnitude (very poor, poor, and fair), attainment status and numbers of parameters ranked as very poor, poor, and fair. Blank cells currently lack data for an attribute.*

| Site ID/ Year | River | River Mile | Drain-age Area (mi.$^2$) | Identified Stressors and Narrative Range | | | Scaled Restorability Score | Aq. Life Use Attainment Status | No. Stressor Exceedances | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | Very Poor | Poor | Fair | | | Very Poor | Poor | Fair |
| SC24 2007 | Addison Creek | 10.5 | 2.0 | Imperv-500m;Urban-WS;Dev-WS;TP; TKN; Substr; Chan; WC Metals; | Imperv-30;Imperv-30C; BOD; QHEI; | | 0 | Non - Poor | 12 | 0 | 0 |
| LD04 2012 | Rock Run Creek | 6.5 | 4.9 | Urban-WS;Dev-WS; TKN; Substr; Conduct; | Low DO; QHEI; Chan; Chloride; | Imperv-500m;Imperv-30;Imperv-30C;TP; BOD; WC Metals; | 4.41 | Non - Poor | 12 | 0 | 1 |
| EB14 2007 | Lacey Creek | 2 | 1.8 | Urban-WS;Dev-WS; TKN; Chloride; TSS; | BOD; QHEI; Chan; Conduct; TDS; | Imperv-500m;TP; Substr; | 6.26 | Non - Poor | 9 | 4 | 0 |
| SC48 2007 | Addison Creek | 2.5 | 18 | Imperv-500m;Urban-WS;Dev-WS; WC Metals; | Chan; | Imperv-30;Imperv-30C;TP; BOD; QHEI; Chloride; | 7.08 | Non - Poor | 6 | 2 | 1 |
| SC28 2007 | Addison Creek | 1.5 | 20 | Imperv-500m;Urban-WS;Dev-WS; WC Metals; | TP; Chan; | Imperv-30C; TKN; BOD; QHEI; Substr; Chloride; | 8.38 | Non - Poor | 6 | 4 | 0 |
| SC06 2007 | Arlington Heights Branch Salt Creek | 4 | 7.7 | Urban-WS;Dev-WS; TKN; BOD; Substr; | Low DO; QHEI; Chan; Chloride; | Imperv-500m;Imperv-30; Conduct; TDS; TSS; | 10.69 | Non - Poor | 9 | 0 | 2 |
| SC26 2007 | Addison Creek | 8 | 5 | Urban-WS;Dev-WS;TP; | Imperv-500m; TKN; BOD; QHEI; Chan; | Substr; | 10.86 | Non - Poor | 6 | 2 | 0 |
| WB14 2012 | Winfield Creek | 3.5 | 5 | Urban-WS;Dev-WS; Low DO; | Imperv-500m; TKN; Chan; Chloride; | TP; BOD; QHEI; Substr; TDS; | 12.91 | Non - Poor | 6 | 4 | 0 |
| SC48 2010 | Addison Creek | 2.5 | 18 | Imperv-500m;Urban-WS;Dev-WS; | QHEI; Chan; | Imperv-30;Imperv-30C; Low DO; Substr; Conduct; Turbidity; Sed. Metals; | 14.24 | Non - Poor | 3 | 2 | 2 |

***Table 15**. IPS sites with the lowest Restorability ranking scores, identified stressors by stressor magnitude (very poor, poor, and fair), attainment status and numbers of parameters ranked as very poor, poor, and fair. Blank cells currently lack data for an attribute.*

| Site ID/ Year | River | River Mile | Drain-age Area (mi.$^2$) | Identified Stressors and Narrative Range | | | Scaled Restorability Score | Aq. Life Use Attainment Status | No. Stressor Exceedances | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | Very Poor | Poor | Fair | | | Very Poor | Poor | Fair |
| SC27 2012 | Addison Creek | 5 | 10 | Imperv-500m;Urban-WS;Imperv-30;Imperv-30C;Dev-WS; | Low DO; Substr; | QHEI; Chan; Conduct; | 15.06 | Non - Poor | 3 | 4 | 1 |
| EB21 2007 | East Branch DuPage River | 20.5 | 14.2 | Urban-WS;Dev-WS;TP; Substr; WC Metals; | QHEI; Chan; | Imperv-500m; Chloride; | 15.67 | Non - Poor | 12 | 0 | 0 |
| EB36 2007 | East Branch DuPage River | 19 | 16 | Urban-WS;Dev-WS; Chan; WC Metals; | TP; BOD; QHEI; Substr; | Imperv-500m; TKN; Chloride; TSS; | 15.76 | Non - Poor | 9 | 2 | 1 |
| SC27 2007 | Addison Creek | 5 | 10 | Imperv-500m;Urban-WS;Imperv-30;Imperv-30C;Dev-WS; WC Metals; | TP; TKN; BOD; Substr; | QHEI; Chan; | 16.07 | Non - Poor | 6 | 4 | 0 |
| SC06 2010 | Arlington Heights Branch Salt Creek | 4 | 7.7 | Urban-WS;Dev-WS; QHEI; Substr; | Low DO; Chan; | Imperv-500m;Imperv-30; Conduct; | 16.35 | Non - Poor | 6 | 2 | 1 |
| SC24 2010 | Addison Creek | 10.5 | 2 | Imperv-500m;Urban-WS;Dev-WS; Substr; Chan; | Imperv-30;Imperv-30C; QHEI; | Low DO; Turbidity; Sed. Metals; | 17.16 | Non - Poor | 6 | 0 | 1 |

to the General Use benchmark. These sites generally had few or no stressors in the very poor and poor categories and most of their identified stressors were in the "fair" category. This array of results also illustrates the reality of missing data in the NE Illinois IPS database. Because data is form different programs with different views of data pairing and sufficiency, gaps will exist. However, care needs to be taken that a lack of identified stressors is not necessarily due to a lack of data. The Power BI Restorability page in the NE Illinois IPS Dashboard has a data confidence rating (0-5) that indicates the sufficiency of the data in each category (i.e., multiple parameters available at that site). For sites with no stressor exceedances (e.g., the Prairie Creek reference sites), it is possible that natural factors, upstream or downstream conditions, or some unmeasured stressor could be contributing to the minor level of impairment that was observed. Because sites at the upper end of the Restorability Rating Score are close to meeting the aquatic life benchmarks they can be seen as "low hanging fruit" where restoration efforts should result in gains in aquatic life condition and attainment. They should also be viewed from a reach perspective to see whether broader scale restoration efforts may be needed (i.e., adjacent impaired sites by similar or different stressors) or whether they are adjacent to attaining sites so that restoration would be paired with efforts to reduce existing threats. This is where the density of monitoring sites is important to water quality management efforts. Where sites are isolated it may be difficult to ascertain whether impairments are the tip of an impairment "iceberg" or simply a localized stressor scenario.

**Low Restorability Ranking Score HUC12 Watersheds and Stream Reaches**

The preceding discussion focused on site-specific Restorability ratings, but here the focus is on the reach and watershed scale Restorability rankings. As was discussed earlier, cumulative stressor impacts tend to reduce populations of sensitive species and taxa at watershed scales such that the fIBI and mIBI may be limited even at sites within a watershed that have lower stressor levels than neighboring sites. Table 16 summarizes, based on the mean Restorability ranking score at HUC12 spatial scales, the least restorable watersheds in the study area with a minimum of five (5) biological sites in a given year. Addison Creek, which was among the least restorable at the site scale, is at the top of the least restorable HUC12 watersheds. The Addison Creek watershed is mostly channel modified with Poor to Very Poor habitat and Poor to Very Poor biological scores, numerous chemical threshold exceedances, and very high IC. Even though Addison Creek has low Restorability ranking scores, it improved incrementally between 2007 and 2016 from a score of 12.5 to 23.4. A low Restorability ranking score does not mean restoration efforts should be discounted, but rather the difficultly of restoration is quantified and may indicate that there is a ceiling below full recovery that can be attained. Other HUC12 watersheds with low Restorability Rating Scores included watersheds in the Salt Creek and the East Branch DuPage River subbasins (Table 16). The upper end of the restorability gradient includes watersheds such as the lower DuPage River mainstem HUC12, some of the upper Des Plaines watersheds such as Mill Creek and Bull Creek and some of the outlying watersheds that were included as part of the historical data incorporated into the NE Illinois IPS from IEPA and other sources (bottom of Table 16). An important caveat with missing stressor data is they were assumed to be neutral (i.e., assigned scores of 4 in the 0.1-10 rating system) such that historical data which oftentimes has limited associated stressor data could underestimate or over-

***Table 16**. IPS sites with the highest Restorability ranking scores, stressor magnitude (Very Poor, Poor, and Fair), attainment status, and the numbers of parameters ranked as Very Poor, Poor, and Fair.*

| Site ID/ Year | Illinois IPS Sites Type | River | River Mile | Drain-age Area (sq. mi) | fIBI | mIBI | Identified Stressors and Narrative Range | | | Scaled Restor-ability Ranking Score | Aq. Life Attain-ment Status | No. Stressor Exceedances | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | Very Poor | Poor | Fair | | | Very Poor | Poor | Fair |
| LD27 2015 | CORE | Tributary #1 | 0.15 | 2.8 | 33* | 42.9 | | | Nitrate; | 97.7 | Partial | 0 | 0 | 1 |
| DZS-01 2009 | SEC | COVEL CREEK | 5.5 | 67.3 | 52 | 34.6* | | TKN; | QHEI; Substr; | 97.64 | Partial | 0 | 2 | 1 |
| FA-06 2010 | REF | PRAIRIE CREEK | 12.3 | 28.1 | 37* | 81.2 | | | | 96.79 | Partial | 0 | 0 | 0 |
| FA-01 2010 | REF | PRAIRIE CREEK | 0.15 | 48.9 | 36* | 72.2 | | | | 95.56 | Partial | 0 | 0 | 0 |
| I-3 2013 | REF | Little Indian Creek | 5.1 | 82.6 | 40.5* | 71.4 | | | | 93.35 | Partial | 0 | 0 | 0 |
| DTA-08 2012 | SEC | INDIAN CREEK | 16 | 125.58 | 39* | 52 | | | Max DO; | 93.13 | Partial | 0 | 0 | 0 |
| 11-1 2016 | CORE | Mill Creek | 0.7 | 63.78 | 31* | 50.7 | | | TKN; QHEI; Substr; Chloride; | 91.71 | Partial | 0 | 0 | 1 |
| 14-1 2016 | CORE | Bull Creek | 0.5 | 11.69 | 36* | 63.4 | | Urban-WS;Dev-WS; Chloride; | Max DO; Conduct; | 91.55 | Partial | 0 | 2 | 1 |
| 13-7 2016 | CORE | Bull's Brook | 0.25 | 2.69 | 26* | 39.9* | | | QHEI; Substr; | 91.07 | Non - Fair | 0 | 0 | 1 |
| 11-2 2016 | CORE | Mill Creek | 1.71 | 62.25 | 32* | 25.8* | | Turbidity; | TKN; Substr; Chloride; TSS; | 90.59 | Non - Fair | 0 | 0 | 1 |
| LD15 2015 | CORE | Lily Cache Creek | 6.3 | 21.4 | 33* | 51.4 | Urban-WS;Dev-WS; | | Imperv-500m; Chloride; | 89.45 | Partial | 3 | 0 | 0 |
| PQCLA-01 2011 | SEC | Union Ditch No 3 | 11 | 58.86 | 36* | 49.1 | | | Max DO; | 89.32 | Partial | 0 | 0 | 0 |
| LD14 2018 | CORE | DuPage River | 26.6 | 204 | 41 | NA | | | | 89.03 | Non - Fair | 0 | 0 | 0 |
| F-2 2012 | REF | Ferson Creek | 7.6 | 11.4 | 25* | 63.6 | | | Urban-WS;Dev-WS; | 88.21 | Partial | 0 | 0 | 1 |

estimate the actual Restorability. A good example of this is seen in the Addison Creek watershed. Complete intensive pollution surveys were conducted by MBI in 2007, 2010, 2013, 2016, and 2021 (not yet included). These data all cluster together in Table 16 at the low end of Restorability, but showed a trend of improvement with the Restorability ranking score improving from 12.5 in 2007 to 24.2 in 2016. A much smaller and incomplete bioassessment in 1987 resulted in a Restorability score of 68.9 that is an artifact of incomplete stressor data. This tool is designed to be used with data generated by watershed level, intensive pollution surveys with paired stressor and biological data. For other watersheds with incomplete monitoring data the results should be interpreted with these limitations and liabilities in mind. Additionally, these gaps should receive a high priority for being filled with more complete data.

Plots of mean HUC12 Restorability ranking scores vs. the HUC12 fIBI, mIBI, QHEI, and QHEI channel scores (watersheds with more than five biological sites in a given year) are illustrated in Figure 58. Not surprisingly there is a strong positive association between these variables since they are partly derived from these scores (Figure 58). Fish IBI and QHEI show a tighter relationship with Restorability than the mIBI and the QHEI channel metric which have a few outliers towards the middle to upper end of the relationships. Some of the variation with lower mIBI scores associated with moderate Restorability scores is likely due to some of the older, incomplete watershed surveys that result in inflated Restorability scores.

The mean HUC12 Restorability shows a threshold relationship with drainage area (Figure 59). The lowest Restorability measures are all associated with small streams, and no larger streams or rivers (e.g., >100 sq. mi) have low Restorability (Figure 58, top left). Despite this tendency in the results, small streams can have high Restorability. Small streams have a greater likelihood of having more serious habitat degradation and are more susceptible being in closer proximity to stormwater impacts (e.g., chlorides, nutrients) that are delivered during storm events. Streams with low Restorability are also more likely to have higher watershed level development (Figure 59, bottom left) and land uses with high IC within spatial buffer areas (Figure 59, bottom right, 500m spatial buffer impervious cover).

## Restorability Ranking Scores for Illinois Stream Reaches (AUIDs)

Tables 17 and 18 are a subset of the HUC12 watersheds (Table 17) and stream reaches (AUIDs, Table 18) with summaries of selected IPS variables within reaches ranked by the lowest Restorability ranking scores (top) and highest Restorability ranking scores (bottom). These least restorable reaches are dominated by streams in the least restorable HUC12 watersheds with many from the Addison Creek watershed (Table 17). These reaches generally have sites that are within the highest IC and are characterized by degraded habitat and multiple stressors in the Very Poor range. Stream reaches ranked as the most restorable (Restorability Scores >60) generally have less severe impairment at sites that are frequently in “partial” attainment, i.e., the fIBI or mIBI meeting their benchmarks while the other is impaired (Table 18). Relatively few stressors are present and where they occur they are largely in the Fair range with minor habitat disturbances (e.g., some excessive silt or embeddedness) or slightly elevated nutrients or ionic strength parameters.

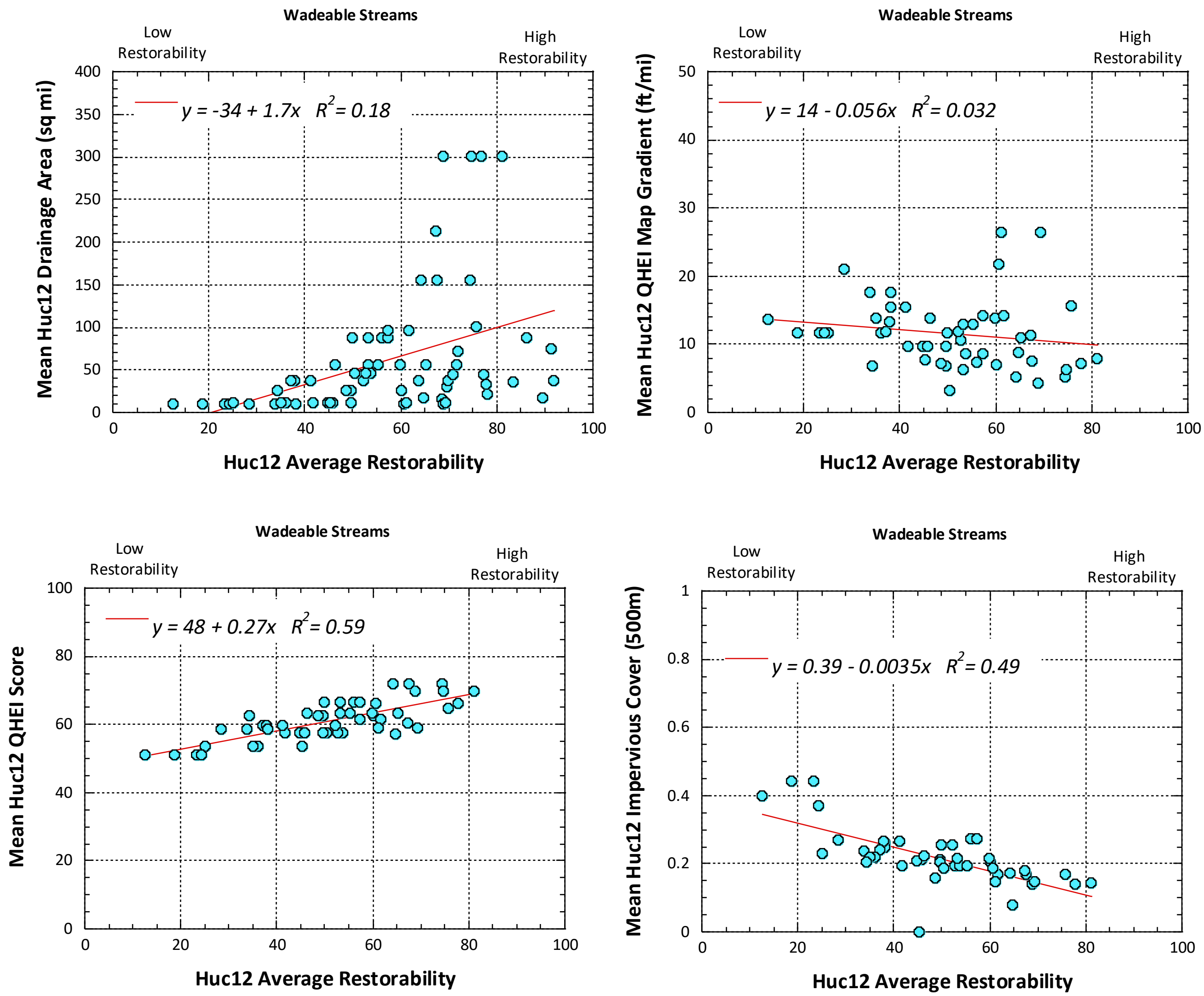



***Figure 58**. Plots of average HUC12 Restorability ranking score vs. mean HUC12 fIBI (top left), mean HUC12 mIBI (top right), mean HUC12 QHEI score (bottom left), and mean HUC12 QHEI channel score (bottom right). Data from HUC12 watersheds that have more than 5 biological sites in a given year.*

The Power BI IPS Dashboard will allow the users to peruse all of the parameters that are available and to "drill down" from the HUC12 watershed scale to the reach and then to the site scale and *visa versa* to explore the influence of cumulative effects or the influence of neighboring reaches and sites. The dense sampling design of the watershed group surveys supports the consideration of cumulative impacts whereas data collected at lower spatial densities is unable to do so. The strong influence of IC and urban land uses, particularly on small streams, can be seen in plots of mean IC in reaches vs. the mean drainage area at sites in each AUID. Figure 60 illustrates this for the 30m clipped spatial buffer data and it is clear that the highest IC are in the smaller streams.

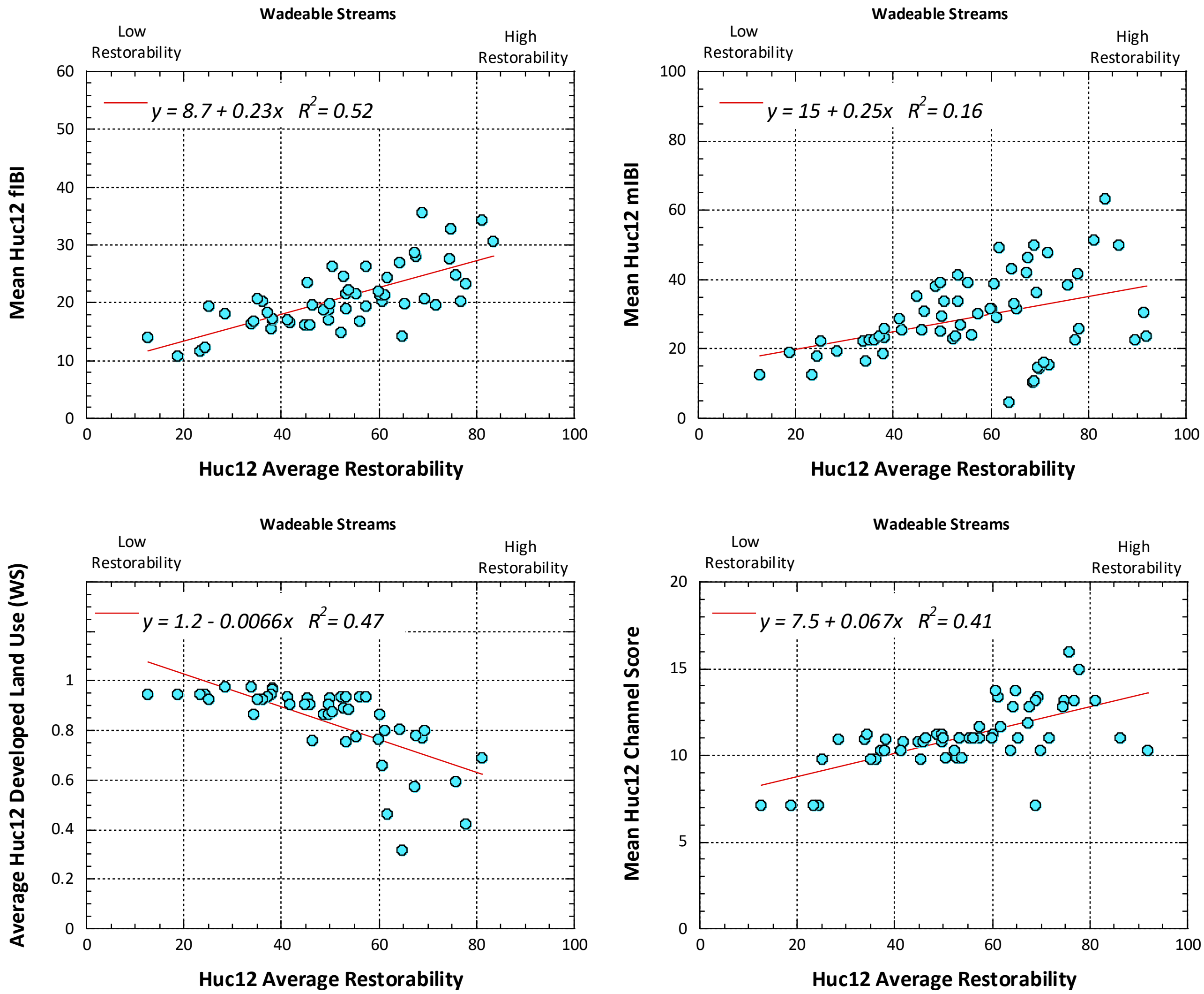



***Figure 59****. Plots of average HUC12 Restorability Ranking Score vs. mean HUC12 drainage area (top left), mean HUC12 map gradient (top right), mean HUC12 developed land uses (WS) (bottom left) and mean HUC12 Impervious land in 500m buffer (bottom right). Data from HUC12 watersheds that have more than 5 biological sites in a given year.*

Figure 61 illustrates the relationship between the Restorability ranking score and the mean AUID fIBI, mIBI, QHEI, and QHEI channel metric. As expected there is a strong positive association. Figure 62 illustrates similar plots for the Restorability ranking score vs. drainage area, map gradient, and two land use variables (developed land uses at a watershed scale and impervious land cover in a 500m spatial buffer. As with the HUC12 and site specific scales, small streams are more vulnerable to having low Restorability scores and larger wadeable streams generally have higher Restorability scores and the likely rationale for this is discussed above.

***Table 17.*** *List of Illinois IPS data summarized by Huc12 watersheds and year for wadeable sites in the NE Illinois IPS study area. Only sites with >5 biological samples are included. Blank cells currently lack data for an attribute.*

| Huc12 Code | Huc12 Name | Year | Drainage Area (sq. mi) | Map Gradient (ft./mi) | Bio Sites | Mean fIBI | fIBI Rank | Mean mIBI | mIBI Rank | Mean Attainment Status | QHEI Sites | Mean QHEI | Mean QHEI Rank | Mean Channel State | Mean Imperv. (WS) | Mean Imperv. 500m | Mean Restor-ability Rating Score |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 071200040403 | Addison Creek | 2007 | 10.8 | 13.7 | 6 | 14.0 | 7.8 | 12.6 | 8.2 | 4 | 24 | 51.1 | 6 | 9.5 | 0.34 | 0.398 | 12.5 |
| 071200040403 | Addison Creek | 2010 | 10.8 | 11.7 | 6 | 10.9 | 8.3 | 19 | 7.3 | 4 | 24 | 51.1 | 6 | 9.5 | 0.387 | 0.443 | 18.8 |
| 071200040403 | Addison Creek | 2016 | 10.8 | 11.7 | 6 | 11.6 | 8.2 | 12.7 | 8.2 | 4 | 24 | 51.1 | 6 | 9.5 | 0.387 | 0.443 | 23.4 |
| 071200040403 | Addison Creek | 2013 | 10.8 | 11.7 | 7 | 12.3 | 8.1 | 18.1 | 7.4 | 4 | 24 | 51.1 | 6 | 9.5 | 0.315 | 0.370 | 24.2 |
| 071200040803 | Headwaters E. Br. Du Page R. | 2007 | 10.9 | 11.7 | 16 | 19.5 | 7 | 22.3 | 6.8 | 4 | 57 | 53.7 | 5.7 | 7.8 | 0.331 | 0.231 | 25 |
| 071200040401 | Upper Salt Creek | 2007 | 9.7 | 20.9 | 12 | 18.1 | 7.2 | 19.4 | 7.2 | 4 | 49 | 58.7 | 5.2 | 7.5 | 0.300 | 0.270 | 28.3 |
| 071200040401 | Upper Salt Creek | 2010 | 9.7 | 17.6 | 12 | 16.4 | 7.5 | 22.3 | 6.8 | 4 | 49 | 58.7 | 5.2 | 7.5 | 0.265 | 0.239 | 33.7 |
| 071200040802 | Middle West Branch Du Page River | 2012 | 26.2 | 6.7 | 12 | 16.9 | 7.4 | 16.5 | 7.6 | 4 | 45 | 62.5 | 4.7 | 7.6 | 0.306 | 0.205 | 34.2 |
| 071200040803 | Headwaters E. Br. DuPage R. | 2014 | 10.9 | 13.8 | 18 | 20.7 | 6.8 | 22.5 | 6.8 | 4 | 57 | 53.7 | 5.7 | 7.8 | 0.311 | 0.221 | 35.1 |
| 071200040803 | Headwaters E. Br. DuPage R. | 2011 | 10.9 | 11.7 | 18 | 20.2 | 6.9 | 22.7 | 6.7 | 4 | 57 | 53.7 | 5.7 | 7.8 | 0.311 | 0.221 | 36.1 |
| 071200040402 | Middle Salt Creek | 2010 | 37.6 | 11.9 | 14 | 18.4 | 7.2 | 23.9 | 6.6 | 4 | 57 | 59.9 | 5.2 | 8 | 0.303 | 0.242 | 37.2 |
| 071200040402 | Middle Salt Creek | 2016 | 37.6 | 13.3 | 12 | 15.5 | 7.6 | 18.7 | 7.3 | 4 | 57 | 59.9 | 5.2 | 8 | 0.326 | 0.267 | 37.9 |
| 071200040401 | Upper Salt Creek | 2013 | 9.7 | 17.6 | 12 | 17.1 | 7.4 | 23.5 | 6.6 | 4 | 49 | 58.7 | 5.2 | 7.5 | 0.300 | 0.262 | 38 |
| 071200040401 | Upper Salt Creek | 2016 | 9.7 | 15.5 | 13 | 17.3 | 7.3 | 25.8 | 6.3 | 4 | 49 | 58.7 | 5.2 | 7.5 | 0.299 | 0.248 | 38.2 |
| 071200040402 | Middle Salt Creek | 2007 | 37.6 | 15.5 | 13 | 17.0 | 7.4 | 28.9 | 5.3 | 4 | 57 | 59.9 | 5.2 | 8 | 0.328 | 0.265 | 41.2 |
| 071200040801 | Upper West Branch Du Page River | 2012 | 11.8 | 9.7 | 16 | 16.7 | 7.4 | 25.4 | 6.2 | 4 | 64 | 57.6 | 5.4 | 7 | 0.276 | 0.194 | 41.6 |
| 071200040801 | Upper West Branch Du Page River | 2006 | 11.8 | 9.7 | 16 | 16.1 | 7.5 | 35.1 | 5.2 | 4 | 64 | 57.6 | 5.4 | 7 | 0.279 | 0.208 | 44.7 |
| 071200040803 | Headwaters E. Br. DuPage R. | 2012 | 10.9 | 7.7 | 7 | 23.5 | 6.4 | | | 3 | 57 | 53.7 | 5.7 | 7.8 | 0 | 0 | 45.4 |
| 071200040801 | Upper West Branch Du Page River | 2009 | 11.8 | 9.7 | 17 | 16.3 | 7.5 | 25.4 | 6.4 | 4 | 64 | 57.6 | 5.4 | 7 | 0.296 | 0.213 | 45.9 |
| 071200040805 | Lower West Branch Du Page River | 2012 | 56.4 | 13.9 | 15 | 19.6 | 7 | 30.9 | 5.4 | 3 | 65 | 63.2 | 4.8 | 7.4 | 0.247 | 0.222 | 46.3 |
| 071200040802 | Middle West Branch Du Page River | 2006 | 26.2 | 7.1 | 10 | 18.8 | 7.1 | 38 | 3.8 | 4 | 45 | 62.5 | 4.7 | 7.6 | 0.224 | 0.158 | 48.7 |
| 071200040801 | Upper W. Br. DuPage R. | 2015 | 11.8 | 9.7 | 16 | 17.1 | 7.4 | 25.2 | 6.3 | 4 | 64 | 57.6 | 5.4 | 7 | 0.296 | 0.213 | 49.7 |
| 071200040802 | Middle W. Br. DuPage R. | 2009 | 26.2 | 6.7 | 11 | 18.7 | 7.1 | 39.4 | 4.2 | 4 | 45 | 62. | 4.7 | 7.6 | 0.306 | 0.205 | 49.7 |
| 071200040404 | Lower Salt Creek | 2007 | 88.4 | 11.6 | 18 | 19.8 | 7 | 29.4 | 5.4 | 3 | 78 | 66.4 | 4.6 | 7.1 | 0.286 | 0.254 | 49.8 |

***Table 17.*** *List of Illinois IPS data summarized by Huc12 watersheds and year for wadeable sites in the NE Illinois IPS study area. Only sites with >5 biological samples are included. Blank cells currently lack data for an attribute.*

| Huc12 Code | Huc12 Name | Year | Drainage Area (sq. mi) | Map Gradient (ft./mi) | Bio Sites | Mean fIBI | fIBI Rank | Mean mIBI | mIBI Rank | Mean Attainment Status | QHEI Sites | Mean QHEI | Mean QHEI Rank | Mean Channel State | Mean Imperv. (WS) | Mean Imperv. 500m | Mean Restor-ability Rating Score |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 071200040804 | E. Br. DuPage R. | 2007 | 46.1 | 3.2 | 12 | 26.3 | 6 | 33.7 | 4.7 | 3 | 50 | 57.7 | 5.4 | 7.7 | 0.253 | 0.186 | 50.5 |
| 071200040402 | Middle Salt Creek | 2013 | 37.6 | 11.9 | 14 | 15.0 | 7.7 | 22.9 | 6.6 | 4 | 57 | 59.9 | 5.2 | 8.0 | 0.298 | 0.255 | 52.2 |
| 071200040804 | E. Br. DuPage R. | 2014 | 46.1 | 10.6 | 20 | 24.7 | 6.2 | 23.7 | 6.5 | 3 | 50 | 57.7 | 5.4 | 7.7 | 0.281 | 0.195 | 52.6 |
| 071200040404 | Lower Salt Creek | 2010 | 88.3 | 6.3 | 19 | 18.9 | 7.1 | 33.7 | 4.9 | 3 | 78 | 66.4 | 4.6 | 7.1 | 0.258 | 0.199 | 53.2 |
| 071200040805 | Lower W. Br. DuPage R. | 2009 | 56.4 | 12.9 | 15 | 21.6 | 6.7 | 41.5 | 4.6 | 3 | 65 | 63.2 | 4.8 | 7.4 | 0.226 | 0.216 | 53.3 |
| 071200040804 | E. Br. DuPage R. | 2011 | 46.1 | 8.7 | 19 | 22.2 | 6.6 | 26.9 | 5.9 | 3 | 50 | 57.7 | 5.4 | 7.7 | 0.278 | 0.196 | 53.6 |
| 071200040805 | Lower W. Br. DuPage R. | 2006 | 56.4 | 12.9 | 16 | 21.5 | 6.7 | 39.1 | 3.9 | 3 | 65 | 63.2 | 4.8 | 7.4 | 0.200 | 0.193 | 55.2 |
| 071200040404 | Lower Salt Creek | 2016 | 88.4 | 7.5 | 19 | 16.8 | 7.4 | 24.2 | 6.2 | 4 | 78 | 66.4 | 4.6 | 7.1 | 0.325 | 0.275 | 55.9 |
| 071200040404 | Lower Salt Creek | 2013 | 88.4 | 8.6 | 21 | 19.5 | 7 | 30.2 | 5.3 | 3 | 78 | 66.4 | 4.6 | 7.1 | 0.324 | 0.273 | 57.2 |
| 071200040808 | Middle DuPage River | 2018 | 96.7 | 14.1 | 8 | 26.4 | 6 | | | 3 | 22 | 61.7 | 5.1 | 6.3 | | | 57.2 |
| 071200040805 | Lower W. Br. DuPage R. | 2015 | 56.4 | 13.8 | 17 | 22.1 | 6.6 | 31.5 | 5 | 3 | 65 | 63.2 | 4.8 | 7.4 | 0.236 | 0.216 | 59.9 |
| 071200040802 | Middle W. Br. DuPage R. | 2015 | 26.2 | 7.0 | 13 | 21.4 | 6.7 | 31.5 | 5.2 | 3 | 45 | 62.5 | 4.7 | 7.6 | 0.306 | 0.205 | 60.0 |
| 071200040502 | Wheeling Drainage Ditch | 2016 | 10.4 | 21.8 | 6 | 20.4 | 6.9 | 39.0 | 3.7 | 3 | 14 | 66.1 | 4.6 | 5.3 | 0.187 | 0.187 | 60.5 |
| 071200040501 | Indian Creek | 2017 | 11.7 | 26.3 | 13 | 21.4 | 6.7 | 29.1 | 5.6 | 3 | 26 | 59.2 | 5.3 | 5.0 | 0.209 | 0.149 | 61.1 |
| 071200040808 | Middle Du Page River | 2015 | 96.7 | 14.1 | 8 | 24.5 | 6.3 | 49.3 | 2.1 | 3 | 22 | 61.7 | 5.1 | 6.3 | 0.158 | 0.170 | 61.6 |
| 071200040402 | Middle Salt Creek | 1987 | 37.6 | | 8 | | | 4.8 | 9.3 | 4 | 57 | 59.9 | 5.2 | 8.0 | | | 63.6 |
| 071200040806 | Upper DuPage River | 2012 | 155.5 | 5.1 | 7 | 27.0 | 5.9 | 43.1 | 3.6 | 3 | 22 | 71.9 | 4 | 5.5 | 0.279 | 0.174 | 64.1 |
| 071200040201 | North Mill Creek | 2016 | 17.4 | 8.8 | 6 | 14.2 | 7.8 | 33.2 | 5.3 | 4 | 7 | 57.1 | 5.5 | 4.7 | 0.100 | 0.078 | 64.6 |
| 071200040805 | Lower W. Br. DuPage R. | 2018 | 56.4 | 11.0 | 10 | 19.8 | 7 | 31.5 | 5.3 | 3 | 65 | 63.2 | 4.8 | 7.4 | | | 65.2 |
| 071200040503 | McDonald Cr.-Des Plaines R. | 2016 | 213.7 | 11.3 | 16 | 28.8 | 5.6 | 42.0 | 3.7 | 3 | 20 | 60.3 | 5.1 | 5.4 | 0.168 | 0.181 | 67.2 |
| 071200040806 | Upper Du Page River | 2015 | 155.5 | 7.6 | 8 | 28.1 | 5.8 | 46.4 | 2.7 | 3 | 22 | 71.9 | 4 | 5.5 | 0.269 | 0.168 | 67.6 |
| 071200060902 | City of Woodstock | 1988 | 15.7 | | 6 | | | 10.4 | 8.5 | 4 | 0 | | | | | | 68.6 |
| 071200040403 | Addison Creek | 1987 | 10.8 | | 7 | | | 10.8 | 8.5 | 4 | 24 | 51.1 | 6 | 9.5 | | | 68.9 |
| 071200040810 | Lower DuPage River | 2012 | 300.4 | 4.3 | 8 | 35.7 | 4.6 | 50 | 2.2 | 2 | 35 | 69.9 | 4 | 5.4 | 0.225 | 0.139 | 68.9 |
| 071200040501 | Indian Creek | 2016 | 11.7 | 26.3 | 13 | 20.8 | 6.8 | 36.5 | 4.7 | 3 | 26 | 59.2 | 5.3 | 5.0 | 0.209 | 0.149 | 69.4 |
| 071200061104 | Flint Creek | 1988 | 30.3 | | 7 | | | 14.8 | 7.9 | 4 | 0 | | | | | | 69.6 |
| 071200040402 | Middle Salt Creek | 1990 | 37.6 | | 7 | | | 14.4 | 7.9 | 4 | 57 | 59.9 | 5.2 | 8.0 | | | 69.9 |
| 070900060202 | City of Huntley-S. Br. Kishwaukee R. | 1990 | 44.1 | | 7 | | | 16.3 | 7.7 | 4 | 0 | | | | | | 70.8 |
| 071200040805 | Lower W. Br. DuPage R. | 1983 | 56.4 | | 6 | 19.7 | 7 | 47.7 | 2.1 | 3 | 65 | 63.2 | 4.8 | 7.4 | | | 71.7 |

***Table 17.*** *List of Illinois IPS data summarized by Huc12 watersheds and year for wadeable sites in the NE Illinois IPS study area. Only sites with >5 biological samples are included. Blank cells currently lack data for an attribute.*

| Huc12 Code | Huc12 Name | Year | Drainage Area (sq. mi) | Map Gradient (ft./mi) | Bio Sites | Mean fIBI | fIBI Rank | Mean mIBI | mIBI Rank | Mean Attainment Status | QHEI Sites | Mean QHEI | Mean QHEI Rank | Mean Channel State | Mean Imperv. (WS) | Mean Imperv. 500m | Mean Restor-ability Rating Score |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 071200040603 | Hickory Creek | 1985 | 71.8 | | 7 | | | 15.5 | 7.8 | 4 | 0 | | | | | | 71.9 |
| 071200040806 | Upper DuPage River | 2018 | 155.5 | 5.3 | 8 | 27.7 | 5.6 | | | 3 | 22 | 71.9 | 4.0 | 5.5 | | | 74.4 |
| 071200040810 | Lower DuPage River | 2018 | 300.4 | 6.3 | 11 | 32.8 | 4.9 | | | 3 | 35 | 69.9 | 4.0 | 5.4 | | | 74.8 |
| 071200040302 | Bull Creek-Des Plaines River | 2016 | 100.9 | 15.6 | 19 | 24.8 | 6.2 | 38.5 | 4.4 | 3 | 20 | 64.7 | 4.6 | 3.8 | 0.172 | 0.168 | 75.7 |
| 071200040810 | Lower DuPage River | 1976 | 300.4 | | 9 | 20.3 | 6.9 | | | 3 | 35 | 69.9 | 4.0 | 5.4 | | | 76.7 |
| 070900060202 | City of Huntley-S. Br. Kishwaukee R. | 2002 | 44.1 | | 8 | | | 22.5 | 6.8 | 3 | 0 | | | | | | 77.2 |
| 071200040202 | Mill Creek | 2016 | 32.7 | 7.2 | 6 | 23.3 | 6.4 | 41.6 | 3.5 | 3 | 8 | 66.4 | 4.3 | 3.1 | 0.152 | 0.140 | 77.8 |
| 071200070401 | Headwaters Somonauk Creek | 1990 | 21.5 | | 7 | | | 25.9 | 6.3 | 3 | 0 | | | | | | 77.9 |
| 071200040810 | Lower DuPage River | 2015 | 300.4 | 7.9 | 11 | 34.3 | 4.8 | 51.3 | 2.2 | 2 | 35 | 69.9 | 4.0 | 5.4 | 0.212 | 0.145 | 81.2 |
| 071200061205 | E Br. Poplar Cr.-Poplar Creek | 2002 | 35.8 | | 6 | 30.7 | 4.2 | 63.5 | 2.4 | 3 | 0 | | | | 0 | 0 | 83.5 |
| 071200040404 | Lower Salt Creek | 1995 | 88.4 | | 10 | | | 50.1 | 2.2 | 1 | 78 | 66.4 | 4.6 | 7.1 | | | 86.2 |
| 071200030301 | Headwaters Plum Creek | 1988 | 17.2 | | 6 | | | 22.7 | 6.6 | 3 | 0 | | | | | | 89.6 |
| 071200070306 | Town of Sandwich-L. Rock Cr. | 1990 | 74.8 | | 6 | | | 30.5 | 5.2 | 3 | 0 | | | | | | 91.2 |
| 071200040402 | Middle Salt Creek | 1995 | 37.6 | | 11 | | | 23.6 | 6.3 | 3 | 57 | 59.9 | 5.2 | 8.0 | | | 91.9 |

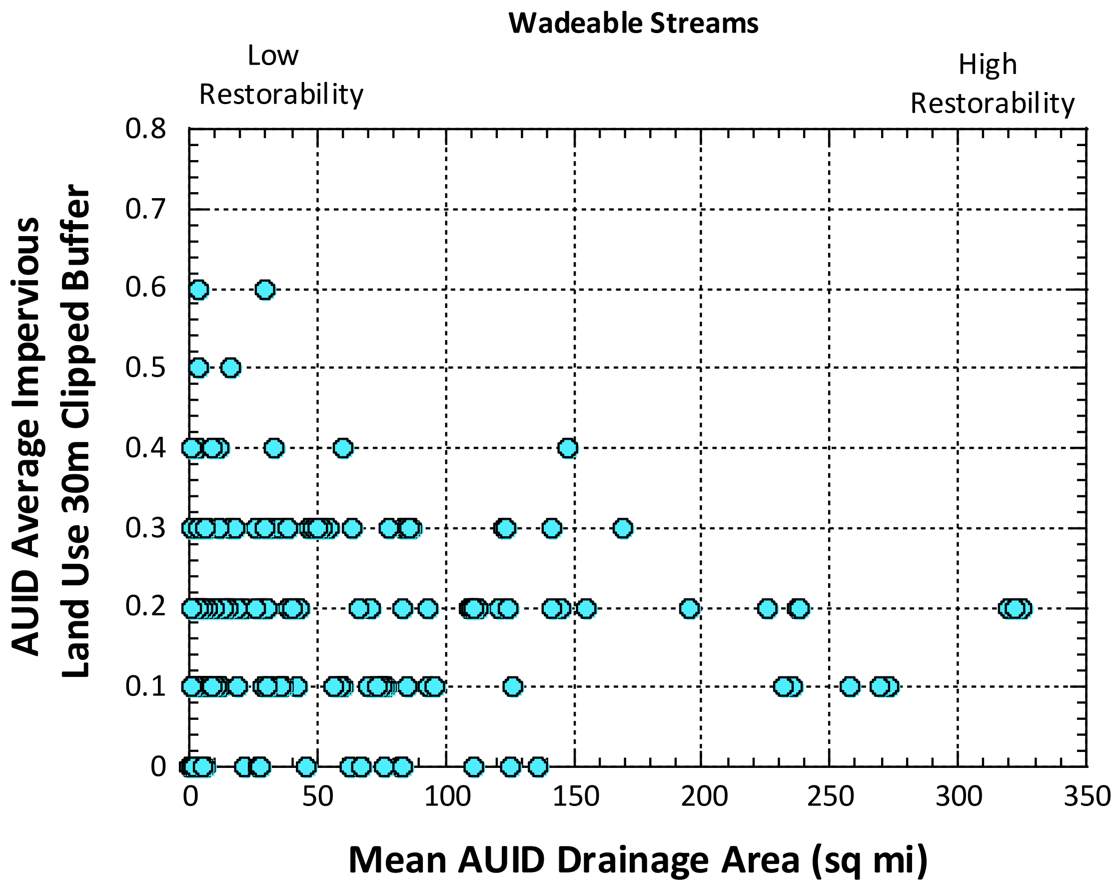



***Figure 60***. *Plot of mean AUID drainage area (sq. mi) vs. mean AUID impervious land cover in a 30m clipped buffer around sites at wadeable streams in the NE Illinois IPS study area.*

### Incorporating External Data into the Analyses of Restorability

The Restorability ranking score is developed on the basis of ambient biological data and performance relative to the Illinois General Use and narrative ranges of condition and ranked ambient stressor and land use data matched to the narrative biological assessment categories. Other types of data, e.g., social, economic, citizen preference, location of parkland and open space, local and regional water management priorities, BMP effectiveness, other priorities such as recreational uses related to human health, can and should be compared to the Restorability, Susceptibility, and Threat scores. The User Manual for the NE Illinois IPS Power BI dashboard (MBI 2022b) has a section that describes how to add additional data to the IPS framework. Smith et al. (2016) discussed the concept of ecological renovation, particularly in urban settings where achieving short-term ecological goals may be unlikely because of the cumulative, watershed-wide scale properties of stressor impacts. They suggest that the integration of social and economic factors in stream restoration efforts can lead to incremental stream protection results that foster a higher probability of long-term ecological recovery. In essence Smith et al. (2016) posit that if projects are designed properly, short-term actions intended to achieve

***Table 18**. List of IPS data summarized by stream reaches (Illinois AUID segments) and year for stream and river sites in the NE Illinois IPS study area. Blank cells currently lack data for an attribute.*

| Illinois AUID Code | AUID Name | Year | Drainage Area (sq. mi) | Bio. Sites | fIBI | fIBI Rank | mIBI | mIBI Rank | QHEI | QHEI Rank | Channel State | Imperv. Landuse (WS) | Imperv. Landuse (500m)` | VP Exceed. | Conductivity (µS/cm) | Ion Rank | TP | Nutrient Rank | Restorability Ranking Score |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Illinois_GLA-02 | Addison Creek (Lower) | 2007 | 16.0 | 3 | 18.0 | 7.2 | 13.2 | 8.2 | 53.3 | 5.8 | 9.8 | 0.4 | 0.5 | 6 | 884 | 3.5 | 1.247 | 6.3 | 10.5 |
| Illinois_GLA-04 | Addison Creek (Upper) | 2007 | 3.5 | 2 | 6.8 | 8.9 | 12 | 8.3 | 47.5 | 6.3 | 9.8 | 0.3 | 0.4 | 9 | 810 | 3 | 3.510 | 9.9 | 10.9 |
| Illinois_GLA-02 | Addison Creek (Lower) | 2010 | 16.0 | 3 | 12.3 | 8.1 | 19.6 | 7.2 | 53.3 | 5.8 | 9.8 | 0.4 | 0.5 | 3 | 1088 | 4.1 | | 6.4 | 15.6 |
| Illinois_GLC | Arlington Heights Branch Salt Creek | 2007 | 10.1 | 3 | 15.5 | 7.6 | 18.9 | 7.3 | 48.7 | 6.2 | 9.0 | 0.3 | 0.2 | 5 | 1400 | 5.8 | 0.103 | 6.6 | 15.8 |
| Illinois_GBLC.2 | Lacey Creek | 2007 | 3.2 | 2 | 17.8 | 7.3 | 20.9 | 7 | 36.9 | 7.1 | 8.8 | 0.3 | 0.2 | 6 | 1258 | 5 | 0.224 | 6.3 | 16.6 |
| Illinois_GLA-04 | Addison Creek (Upper) | 2010 | 3.5 | 2 | 6.0 | 9 | 19.6 | 7.2 | 47.5 | 6.3 | 9.8 | 0.3 | 0.4 | 4.5 | 1035 | 4 | | 4.2 | 18.0 |
| Illinois_GBL-08 | East Branch DuPage River (Middle) | 2007 | 15.1 | 4 | 19.0 | 7.1 | 23.6 | 6.6 | 49.5 | 6.1 | 8.8 | 0.2 | 0.2 | 9.8 | 940 | 3.4 | 2.056 | 8.6 | 18.3 |
| Illinois_GBAA-01 | Rock Run Creek | 2012 | 7.0 | 3 | 13.3 | 7.9 | 12.5 | 8.2 | 49.4 | 6.1 | 4.0 | 0.3 | 0.2 | 9 | 3018 | 9.8 | 1.147 | 7.8 | 19.4 |
| Illinois_GLA-04.1 | Trib. to Addison Creek @ RM 10.35 | 2007 | 1.0 | 1 | 16.5 | 7.4 | | | 51.9 | 5.9 | 8.0 | 0.2 | 0.2 | 6 | 475 | 1.8 | 0.270 | 8.0 | 20.2 |
| Illinois_GLA-02 | Addison Creek (Lower) | 2016 | 16.0 | 3 | 13.3 | 7.9 | 9.5 | 8.7 | 53.3 | 5.8 | 9.8 | 0.4 | 0.5 | 3 | | | | | 21.2 |
| Illinois_GLA-04 | Addison Creek (Upper) | 2013 | 3.5 | 2 | 9.3 | 8.5 | 20.4 | 7.1 | 47.5 | 6.3 | 9.8 | 0.3 | 0.4 | 3 | | | | | 22.0 |
| Illinois_GLA-02 | Addison Creek (Lower) | 2013 | 16.4 | 4 | 12.5 | 8 | 16.7 | 7.6 | 53.3 | 5.8 | 9.8 | 0.4 | 0.5 | 3.8 | 1466 | 5.7 | 0.985 | 5.7 | 23.3 |
| Illinois_GBKF-01 | Winfield Creek | 2012 | 5.3 | 3 | 18.0 | 7.2 | 14.8 | 7.9 | 50.8 | 6 | 9.1 | 0.3 | 0.3 | 6 | 984 | 6.2 | 0.251 | 7.5 | 23.6 |
| Illinois_BF-01 | Salt Creek (HW2) | 2007 | 2.5 | 1 | 17.5 | 7.3 | 22.6 | 6.8 | 71.5 | 4.2 | 7.3 | | | 6 | 1450 | 5.6 | 0.111 | 8.0 | 23.7 |
| Illinois_GLA-04 | Addison Creek (Upper) | 2016 | 3.5 | 2 | 5.8 | 9 | 17.9 | 7.4 | 47.5 | 6.3 | 9.8 | 0.3 | 0.4 | 4.5 | | | | | 23.8 |

***Table 18**. List of IPS data summarized by stream reaches (Illinois AUID segments) and year for stream and river sites in the NE Illinois IPS study area. Blank cells currently lack data for an attribute.*

| Illinois AUID Code | AUID Name | Year | Drainage Area (sq. mi) | Bio. Sites | fIBI | fIBI Rank | mIBI | mIBI Rank | QHEI | QHEI Rank | Channel State | Imperv. Landuse (WS) | Imperv. Landuse (500m)` | VP Exceed. | Conductivity (µS/cm) | Ion Rank | TP | Nutrient Rank | Restorability Ranking Score |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| ***Most Restorable Reaches*** | | | | | | | | | | | | | | | | | | | |
| Illinois_GL-10 | Salt Creek (Upper) | 1995 | 55.0 | 5 | | | 29.9 | 4.7 | 67.4 | 4.6 | 7.3 | | | 3 | | | | | 89.6 |
| Illinois_FBA | Jordan Creek | 2004 | 15.2 | 1 | 37.0 | 4.4 | | | | | | | | 0 | | | | | 91.1 |
| Illinois_DSB-01 | Otter Creek | 1990 | | 2 | 37.5 | 4.2 | 80.7 | 1.7 | 70.5 | 4.4 | | | | 0 | | | | | 91.6 |
| Illinois_HBE-02 | Plum Creek | 1988 | | 2 | | | 37.4 | 4.1 | | | | | | 0 | | | | | 91.6 |
| Illinois_DSA-02 | Bailey Creek | 2009 | 28.4 | 1 | 26.0 | 6.0 | 51.6 | 2.2 | 80.0 | 3 | | | | 0 | 693 | 1.9 | 0.054 | 5.3 | 92.1 |
| Illinois_PQ-14 | Kishwaukee River (Lower) | 2011 | 578 | 2 | 33.0 | 5.0 | 74.6 | 2.1 | 70.0 | 3.5 | 2.0 | 0.1 | 0.3 | 3 | 799 | 2.4 | 0.129 | 2.2 | 92.2 |
| Illinois_PT | S. Kinnikinnick Cr. | 2003 | 9.7 | 1 | 30.0 | 5.5 | 46.4 | 2.1 | | | | | | 0 | 641 | 1.7 | | 2.0 | 92.2 |
| Illinois_DTF-02 | Ferson Creek | 2012 | 38.3 | 3 | 37.7 | 3.6 | 60.5 | 2.3 | 80.1 | 2.9 | 3.5 | 0.1 | 0.1 | 0 | 1151 | 4.4 | 0.083 | 1.6 | 92.4 |
| Illinois_DTA-05 | Indian Creek | 2012 | 125.6 | 1 | 39.0 | 4.1 | 52.0 | 2.2 | | | | 0 | 0 | 0 | 579 | 1.7 | 0.077 | 2.9 | 93.1 |
| Illinois_DSF-01 | Long Point Creek | 1990 | | 4 | | | 28.3 | 5.4 | | | | | | 0 | | | | | 93.3 |
| Illinois_FA-01 | Prairie Creek | 2000 | 48.9 | 1 | 36.0 | 4.6 | | | 82.0 | 2.6 | 5.0 | 0 | 0 | 0 | 708 | 2.6 | 0.03 | 1.3 | 94.6 |
| Illinois_DTKA-04 | N. Br. Nippersink Cr. | 2012 | 67.6 | 1 | 38.0 | 4.3 | 78.0 | 1.8 | | | | 0 | 0 | 0 | 873 | 2.9 | 0.089 | 1.7 | 95.0 |
| Illinois_FB-01 | Forked Creek | 2000 | 102.2 | 1 | 39.0 | 4.1 | | | 77.8 | 3.2 | 5.0 | | | 0 | 590 | 1.6 | 0.020 | 1.4 | 96.1 |
| Illinois_DTB-01 | Somonauk Cr. (Lower) | 1989 | | 5 | | | 32.2 | 4.7 | | | | | | 0 | | | | | 112.0 |
| Illinois_FA-01 | Prairie Creek | 2010 | 38.5 | 2 | 36.5 | 4.5 | 76.7 | 2.1 | 82.0 | 2.6 | 5.0 | 0 | 0 | 0 | 654 | 1.8 | 0.066 | 1.7 | 112.0 |
| Illinois_GB-01.1 | Tributary #1 | 2015 | 2.8 | 1 | 33.0 | 5.0 | 42.9 | 2 | 78.0 | 3.5 | 2.0 | 0.1 | 0 | 0 | 741 | 2.0 | 0.058 | 4.1 | 112.0 |

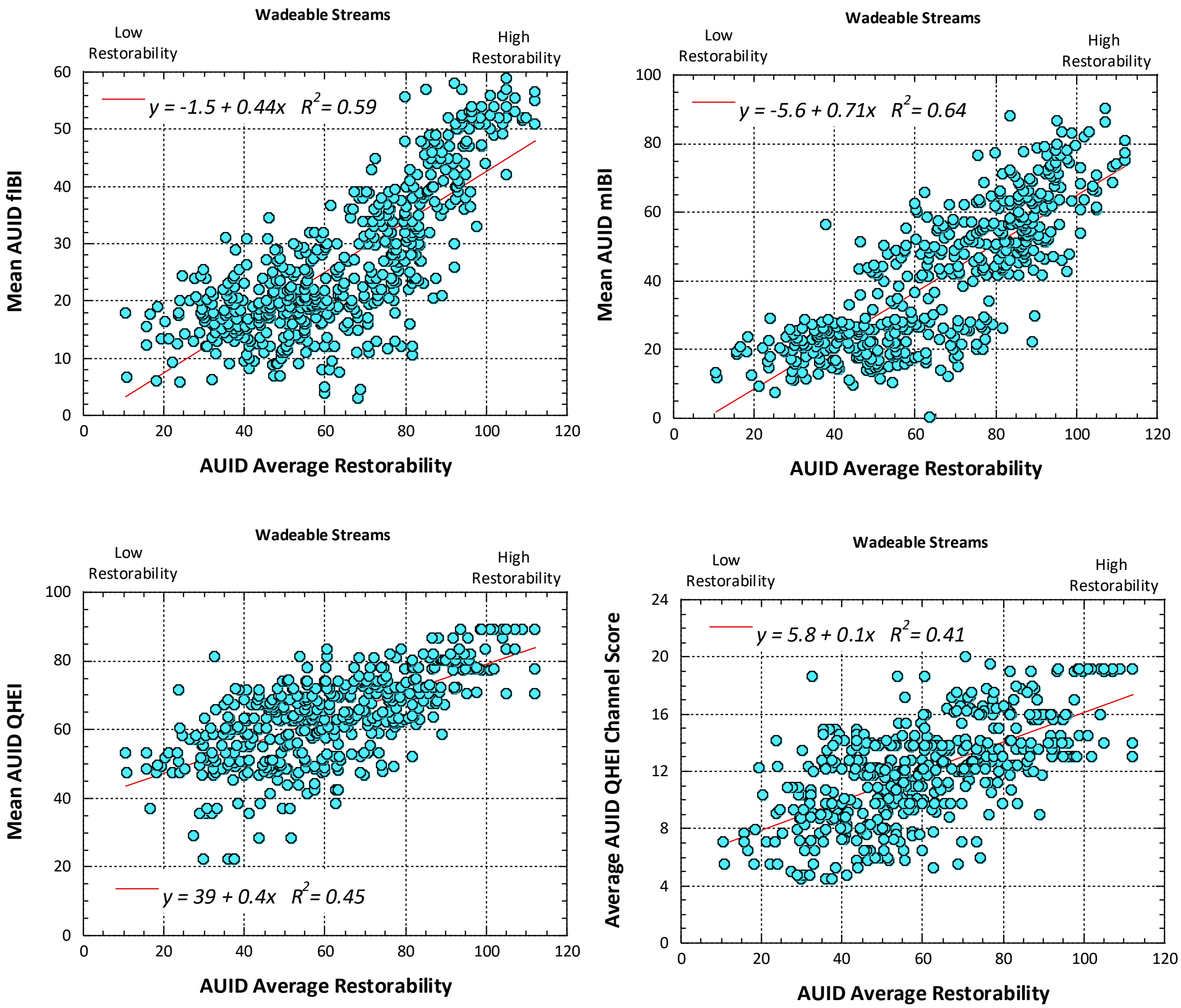



***Figure 61**. Plots of average AUID Restorability Ranking Score vs. mean AUID fIBI (top left), mean AUID mIBI (top right), mean AUID QHEI score (bottom left) and mean AUID QHEI Channel score (bottom right) for stream and river reaches in the NE Illinois IPS study area.*

societal benefits can result in increased public support for actions to achieve long-term ecological improvements in urban streams. When variables such as social measures or demographic data are generated they can be plotted vs. biological measures to determine correlations between such factors which is a key challenge posed by Walsh et al. (2005). Angermeier et al. (2021) found meaningful relationships between stream health, human well-being, and demographics in Virginia using a much coarser dataset than what is available from the IPS study area.

Other than in tabular form, data can be examined graphically using a "bubble plot." In Figure 63 two axes are plotted, such as fIBI vs. mIBI and the point size varies by a third variable, in this case restorability. Reaches are identified on each plot and in Power BI other data can be viewed

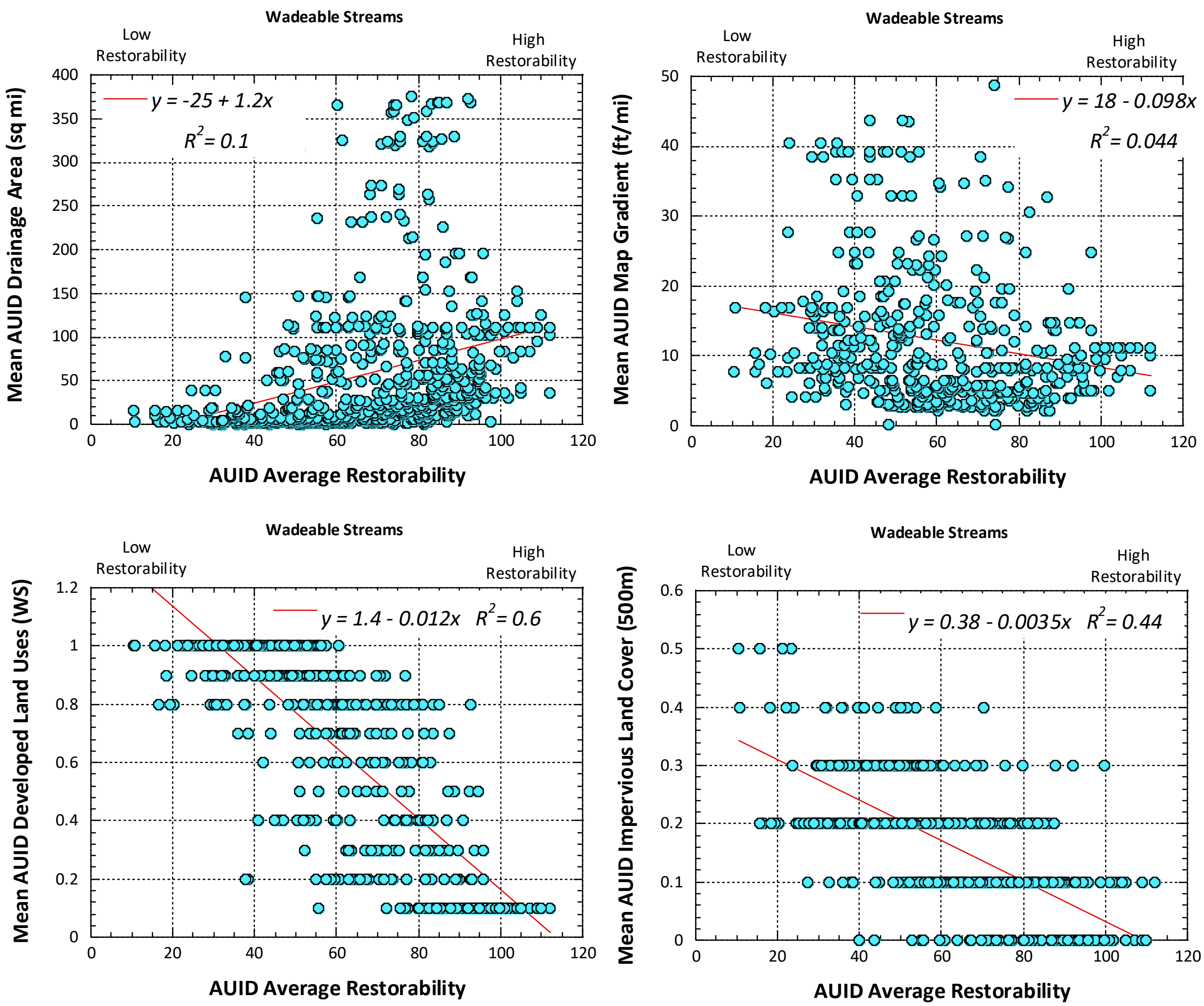



***Figure 62**. Plots of average AUID Restorability Ranking Score vs. mean AUID drainage area (top left), mean AUID map gradient (top right), mean AUID developed land uses (WS) (bottom left) and mean AUID Impervious land in a 500m spatial buffer (bottom right).*

as well. In this plot the most restorable reaches are larger, more to the right and are at the blue end of the color spectrum. The least restorable points are to the left and in red. If one enters another data column such as adjacent parkland or open land adjacent to a stream points that are most restorable can be viewed and grouped together.

## Susceptibility

A Susceptibility ranking was calculated for sites that are meeting the Illinois General Aquatic Life Use biological thresholds with higher scores indicating a greater susceptibility or vulnerability to future degradation and decline. The concept relies on the fact that sensitive fish species and macroinvertebrate taxa decline in a linear manner with increased stress. Highly diverse aquatic assemblages in NE Illinois are rare and are not found in most of the developed

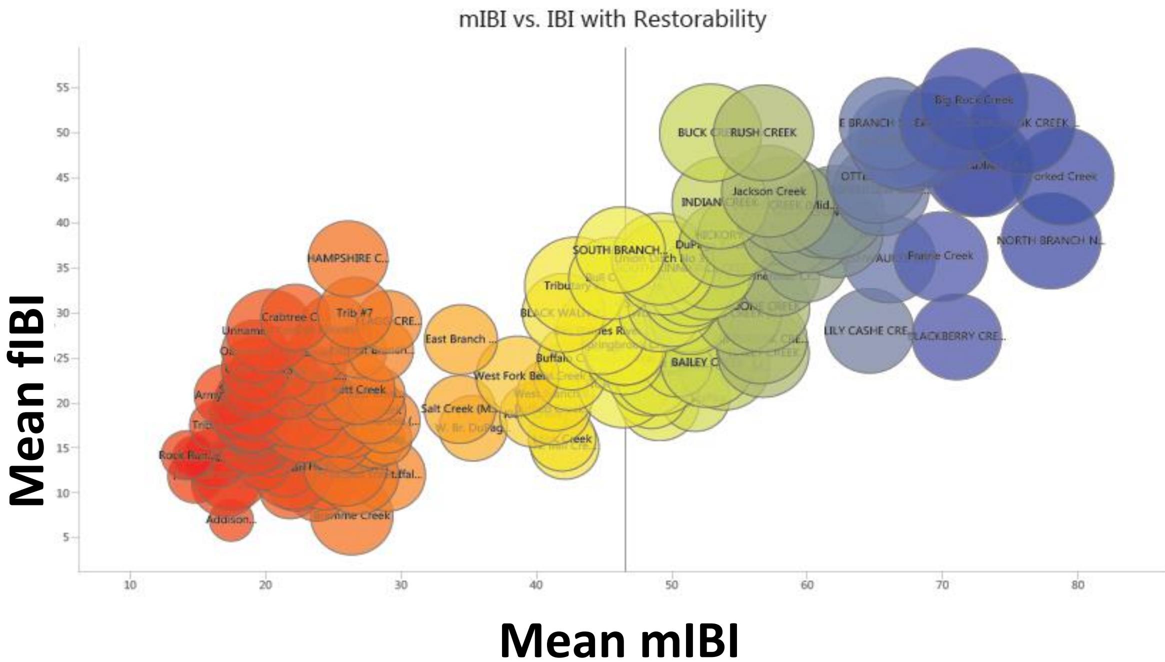



***Figure 63**. Plots of average AUID mIBI vs. mean AUID fIBI with point size and color representing the mean AUID restorability scores; reach name is listed on each point.*

watersheds. Thus sites and reaches with the highest diversity and concentrations of intolerant species were considered to be susceptible to degradation because sites with a high stressor level across multiple stressor categories either have reduced or eliminated populations of their intolerant species or taxa.. In Illinois for example, DeWalt et al. (2009) documented the historical decline and imperilment of stoneflies in Illinois with nearly 28.6% of stonefly species being extinct, extirpate, or in serious decline in the Northeast Morainal region (which includes portions of the NE Illinois IPS study area).

In deriving the Susceptibility score, the most biologically sensitive sites are considered the most at risk or most vulnerable to further increases in stressors. Data from across the Midwest indicates that such waters have been adversely affected by the range of stressors associated with human activities and impacts. Sites that would historically rank as the highest quality and the most susceptible (i.e., with the highest susceptibility scores) are less common. Many of the reference sites outside of the core IPS watersheds where the largest populations of sensitive species/taxa regularly occur have fewer and less severe stressors and lower watershed development.

Sites that are only marginally attaining the General Use aquatic life IBI thresholds and which have a low background level of stressors are considered to have a lower susceptibility. An examination of the Threat scores (see section below) indicates that such sites have existing threats, which means some stressors are elevated above the General Use thresholds. The expected composition of species in streams with a lower Susceptibility score tends to be more resilient to increasing stress and they may naturally lack the most intolerant species that disappear when stressors first increase. The algorithm for determining the Susceptibility score is similar to that of the Restorability score. Sites that have Excellent biological assemblages have higher biological index scores, good spatial buffer land uses, and excellent instream stressor levels will receive the highest Susceptibility scores (>50). There is a similarity among several attributes within the Restorability and Susceptibility ranking algorithms with a slightly higher weighting given to natural channels and sites with more natural buffers in the latter.

## Threat Score

In addition to the Susceptibility ranking, a Threat ranking was developed that focuses on attaining sites with stressors that either exceed or are close to exceeding Excellent and Good thresholds. It also emphasizes stressors that are considered to be more readily controllable. For example, of the eight factors land use is the most difficult to abate. Some of its related stressors may be controllable, but results would take longer to see (e.g., PAHs). The Threat factors and their weighting are depicted in Table 19. Each stressor received a 1 if the stressor was in the fair range, a score of 3 if the stressor was in the poor range, and a score of 7 if the stressor was in the very poor range. The threat score was then normalized to a scale of 0-100 with 0 indicating no known threat (or lack of data) and the highest threat score indicating the presence of multiple stressors ranked poor or very poor.

***Table 19.*** *Scoring components of the Illinois IPS Threat Ranking Score*

| Stressor Category | Max. Weighted Category Rank | Threat Score Component |
|---|---|---|
| **Nutrients** | ≥8 | 7 |
| | ≥6, <8 | 3 |
| | >4, <6 | 1 |
| **Ammonia** | ≥8 | 7 |
| | ≥6, <8 | 3 |
| | >4, <6 | 1 |
| **Habitat** | ≥8 | 7 |
| | ≥6, <8 | 3 |
| | >4, <6 | 1 |
| **PAHs** | ≥8 | 7 |
| | ≥6, <8 | 3 |
| | >4, <6 | 1 |
| **Ions** | ≥8 | 7 |
| | ≥6, <8 | 3 |
| | >4, <6 | 1 |
| **Suspended Sediments** | ≥8 | 7 |
| | ≥6,<8 | 3 |
| | >4, <6 | 1 |
| **Land Use Parameters** | ≥8 | 7 |
| | ≥6, <8 | 3 |
| | >4, <6 | 1 |
| **Water Column Metals** | ≥8 | 7 |
| | ≥6, <8 | 3 |
| | >4, <6 | 1 |

The Threat score can be used to identify sites that currently attain their biological threshold, but which have levels of stressors that if increased could result in a biological impairment. For example, a site may have a low Susceptibility score because it is a General Use designated stream that is marginally attaining the fIBI or mIBI threshold, but which receives a high Threat score because of elevated chemical stressors. The importance of the Susceptibility and Threat rankings is for taking action before impairment occurs, thus it is a protective mode of management that should complement a Restoration focus.

## Comparison with Earlier IPS Data

MBI (Miltner et al. 2011) previously developed an earlier and smaller scale IPS framework using a much smaller database focused entirely in the Upper DuPage River and Salt Creek watersheds. The NE Illinois IPS is built upon what was learned in the previous IPS and subsequent IPS development with a larger database in SW Ohio

(MBI 2015). The goal of the original IPS was ". . . the development a framework for an active biological stressor prioritization system to support a quantitative decision-making process for developing restoration options for impaired reaches of streams and rivers in the DuPage and Salt Creek watersheds." As with the first effort, a major goal of the current NE Illinois IPS is to ". . . correctly and comprehensively identify the sources of stress and impairments." The first IPS was based on *". . . a data set containing over 100 locations [years 2006-2009] with matched chemical, physical and biological data that allowed for analysis and identification of the most proximate suite of stressors."* The current effort resulted in the addition of ten more years of data (up to 2018) in the DuPage River watershed, the addition of data from the Upper Des Plaines watershed, and historical data that included a much wider spatial area in NE Illinois, and the addition of historical data from these same areas totaling ~1130 sites.

The Restorability ranking scores derived for the NE Illinois IPS was calculated in a broader manner than the original IPS which used an algorithm *". . . based on percentile ranks of the number of identified stressors, magnitudes of biological departures, and the amount of open space adjacent to the reach."* One of the problems when using multivariate statistics and regression and classification trees is that although they identify the most widely occurring stressors responsible for biological impairment, the separation of highly correlated variables is an artifact of the sites used in the analyses and the stressor parameters that were available. The previous IPS work (MBI 2010) identified the importance of habitat, land use, chloride and dissolved solids, ammonia-N, and organic enrichment/nutrients using a variety of statistical methods (e.g., cluster analyses, non-metric multidimensional scaling (NMDS), regression trees, and structural equation models. The original IPS analysis produced *". . . nine of the environmental variables identified as being the most likely proximate controlling variables"* (Table 20) and the site Restorability rating was based partially on exceedances of these thresholds.

***Table 20.*** *Environmental thresholds for the most meaningful stressor parameters identified by quantile regression in the original IPS study (Miltner et al. 2011) and the thresholds in the 2020 IPS update.*

| Stressor Parameter | mIBI | fIBI | New IPS Threshold |
|---|---|---|---|
| **Riparian Score** | 5 | Continuous | 6.0 (Fish) |
| **Riffle Score** | 4 | 3 | 5.88 (Fish) |
| **Channel Score** | Continuous | 10 | 14 (Fish) |
| **Substrate Score** | 9 | Continuous | 15 (Fish) |
| **Pool Score** | 7 | 7 | 10 (Fish) |
| **Chloride** | 141 mg/l | 112 mg/l | 120 (Fish) |
| **TKN** | Continuous | 1.0 mg/l | 1.12 (Mac.) |
| **BOD** | Continuous | Continuous | 2.35 (Fish) |
| **$NH_3$-N** | Continuous | 0.15 mg/l | 0.10 (Mac.) |

The 2022 NE Illinois IPS started from the analyses used in the original 2010 effort, but included what was learned from a more data rich IPS effort developed for SW Ohio (MBI 2015). The same stressor-specific sensitive fish species and macroinvertebrate taxa approach (SSD) are used herein as response variables that respond more strongly to individual stressors than the summary indices or their metrics. The results are then linked back to the established fIBI and mIBI thresholds set by IEPA for attainment of the General Use biocriteria, plus benchmarks for five narratives from Excellent to Very Poor biological condition. Using this methodology, thresholds for many more parameters were derived with the addition of a FIT coefficient that compares existing stressor ranks to “back casting”, which compares predicted stressor ranks to existing stressor ranks based on the richness of stressor-specific fish species or macro-invertebrate taxa. The goodness-of-fit (FIT) was ranked for each individual stressor before calculating the a Restorability ranking score for each site.

A Random Forest (RF) regression and classification approach was also used to compare the “univariate” species/taxa based stressor thresholds to the parameter “importance” ranking from the RF models. It was expected that the specific output of key variables in a classification or regression tree would vary with the sites and the parameters used in the analyses. The product of the methodology is list of candidate parameters that contribute to observed aquatic life impairment and threats to attainment.

In comparing the original nine parameters with the newly generated thresholds (Table 20) the habitat thresholds in the original analysis are lower than the newly derived thresholds, but the chemical thresholds for chloride, TKN and ammonia are similar. The new IPS effort leveraged substantially more reference type data for generating thresholds, than did the earlier analyses, and the core of the DuPage River data was relatively degraded. These factors may have contributed to the difference in thresholds. The thresholds were directly derived from fIBI and mIBI relationships with the stressors whereas the newer approach included a step that used stressor sensitive fish and macroinvertebrate taxa to first derive a relationship with the stressor and then back-calculated thresholds and then linked the sensitive species and taxa back to the mIBI and fIBI General Aquatic Life Use and Excellent benchmarks. Plots of sensitive species/taxa vs. a stressor are nearly always more tight than plots with the stressor and indices (see plots in Appendix C).

Figure 64 is a scatter plot of the current NE Illinois IPS Restorability Ranking score vs the original priority ranking score. The correlation is significant, but there is some scatter, probably related to the difference in the algorithms and the weighting of components. The earlier system varied between 1 and 6 whereas the new ranking is continuous between 0 and 100. The lack of higher quality sites in the earlier effort (Figure 64) is evident in the truncation of the original IPS to a maximum score of 60 as all of these original reaches were degraded.

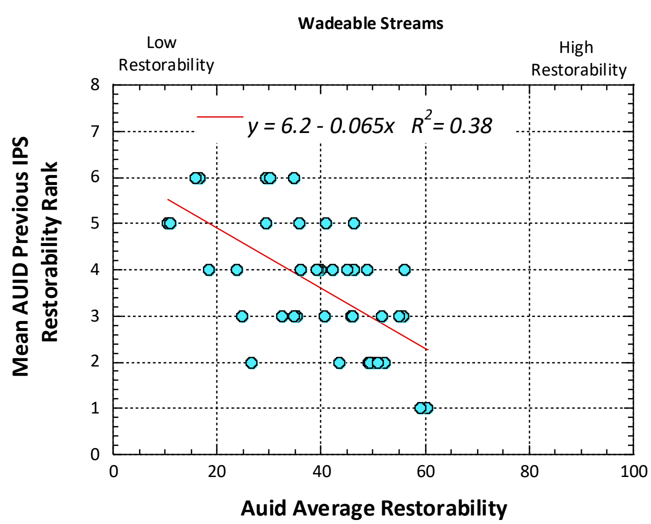



***Figure 64**. Scatter plot of the current Restorability Ranking Score vs. the earlier priority ranking score by AUID (by reach) for common reaches sampled primarily in the DuPage and Salt Creek watersheds.*

The use of the thresholds and restorability data is designed to help select BMPs and to guide water quality management efforts through active adaptive management. The thresholds are not designed to be used as “stand alone” criteria or standards, but rather to inform the selection of the most promising and cost-effective BMPs.

The NE Illinois IPS is more robust partly because of the accrual of watershed intensive pollution survey data that quantifies changes over time as water quality abatement projects are implemented. Each cycle of new data should be used to refine the thresholds and stressors identified as responsible for impairments. The IPS is only as good as the data and analyses that it contains. Several developmental needs have already been identified to better refine the role of nutrients in aquatic life impairment, including chlorophyll a and continuous D.O. data. Refined measures of floodplain function and availability would be helpful in planning and to better understand the role of adjacent land uses in protecting, moderating, or worsening stormwater runoff. For example, the development of a floodplain index and metrics could provide field validation of GIS derived floodplain measures used in the NE Illinois IPS. Outputs of refined stormwater models (e.g., daily flow values at sites) could also be useful within the IPS framework to better discriminate direct flow effects from chemical effects related to impervious land cover measures.

The user oriented module of the NE Illinois IPS is a Power BI dashboard that allows users to explore the data and assessments more effectively. An advantage of Power BI (and similar platforms) is the ability to add new visualizations over time that should enhance product utility. For example, the early mapping tools are relatively basic, but custom ARC MAP coverages can be created and linked to the IPS Power BI dashboard to improve its visual and analytic capabilities. As data is added “time” vectors can be added to graphs and maps to show trends over time. To make the IPS more citizen friendly, the biological index score can be used to identify “indicator taxa” that show attainment of aquatic goals in a more “charismatic manner.” IPS analyses identified rainbow darter (Figure 65) and black redhorse as two fish species associated with good-excellent biological conditions. Such species could provide more “attractive” endpoints for communicating ecological endpoints. Finally, the IPS can be revised over time to include more demographic data to begin to overlay biological condition with demographic data relative to economic and social condition.

***Figure 65**. Rainbow Darter (Etheostoma caeruleum, upper) and Black Redhorse (Moxostoma duquesnei, lower) are two intolerant fish species that can serve as indicators of "good" and "excellent" ecological conditions in the NE Illinois IPS study area.*

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# Appendix A: Watershed Biological and Water Quality Assessments and Other Technical Documents Completed by the NE Illinois Watershed Groups

**DuPage River Salt Creek Watershed Workgroup (DRSCW)**
**Lower DuPage River Watershed Coalition (LDRWC)**
**Des Plaines Watershed Workgroup (DRWW)**
**North Branch Chicago River Watershed Workgroup (NBWW)**

## DuPage River Salt Creek Watershed Workgroup (DRSCW)

Midwest Biodiversity Institute (MBI). 2023. Biological and Water Quality Study of the West Branch DuPage River Watershed 2020. DuPage County, Illinois. Technical Report MBI/2017-8-8. Columbus, OH 43221-0561. 81 pp.

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**Lower DuPage River Watershed Coalition**

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**Des Plaines Watershed Workgroup (DRWW)**

Midwest Biodiversity Institute (MBI). 2022. Biological and Water Quality Assessment of Upper Des Plaines River: 2020. Lake County, Illinois. Technical Report MBI/2022-3-1. Columbus, OH 43221-0561. 75 pp. + appendices.

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**North Branch Chicago River Watershed Workgroup (NBWW)**

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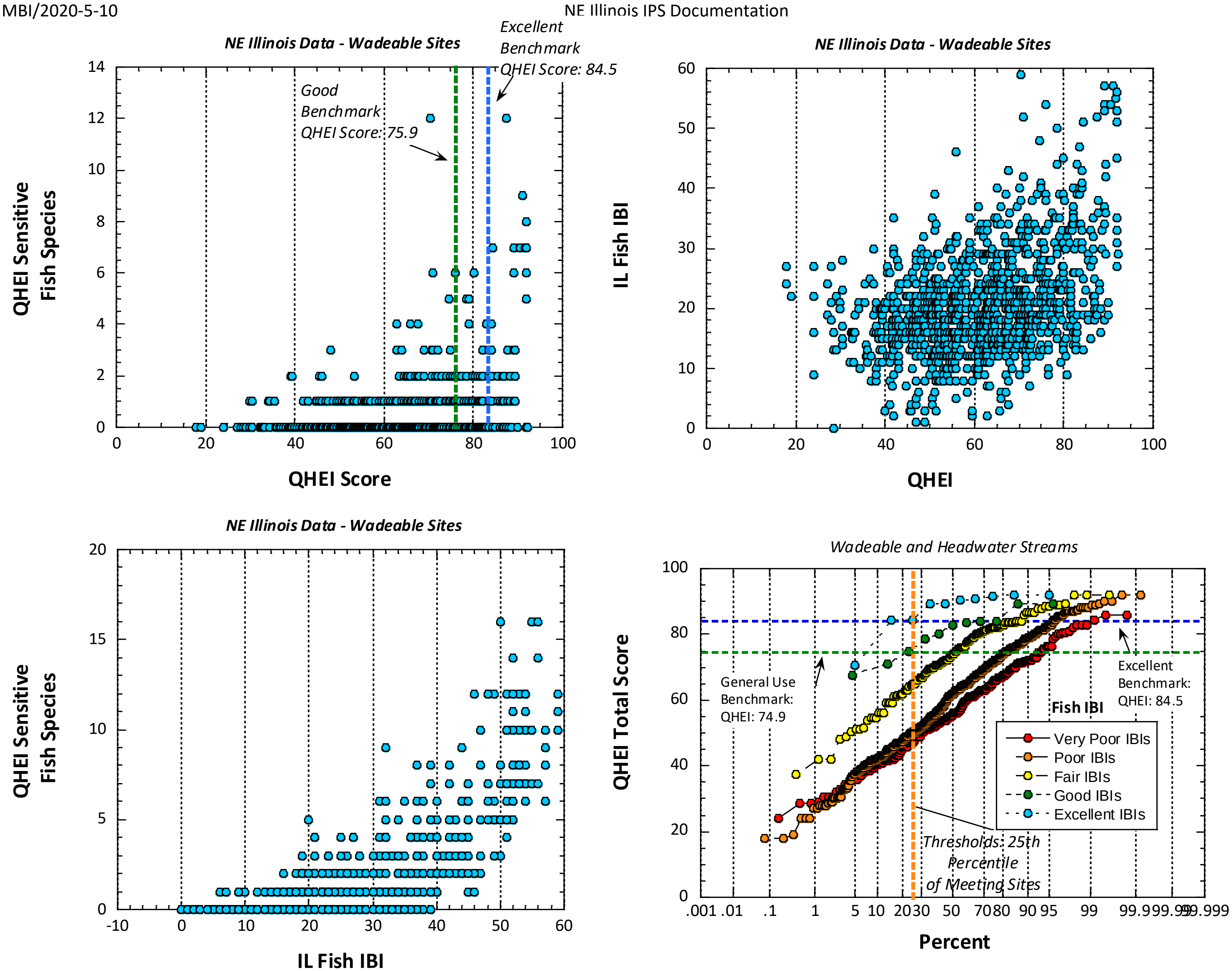
# Appendix B. Gallery of Graphs and Plots of Stressor Parameter Effect Thresholds Based on Fish and Macroinvertebrate Assemblage Responses for the NE Illinois IPS Streams and Rivers Draining <350 Square Miles (mi.$^2$)

## Description

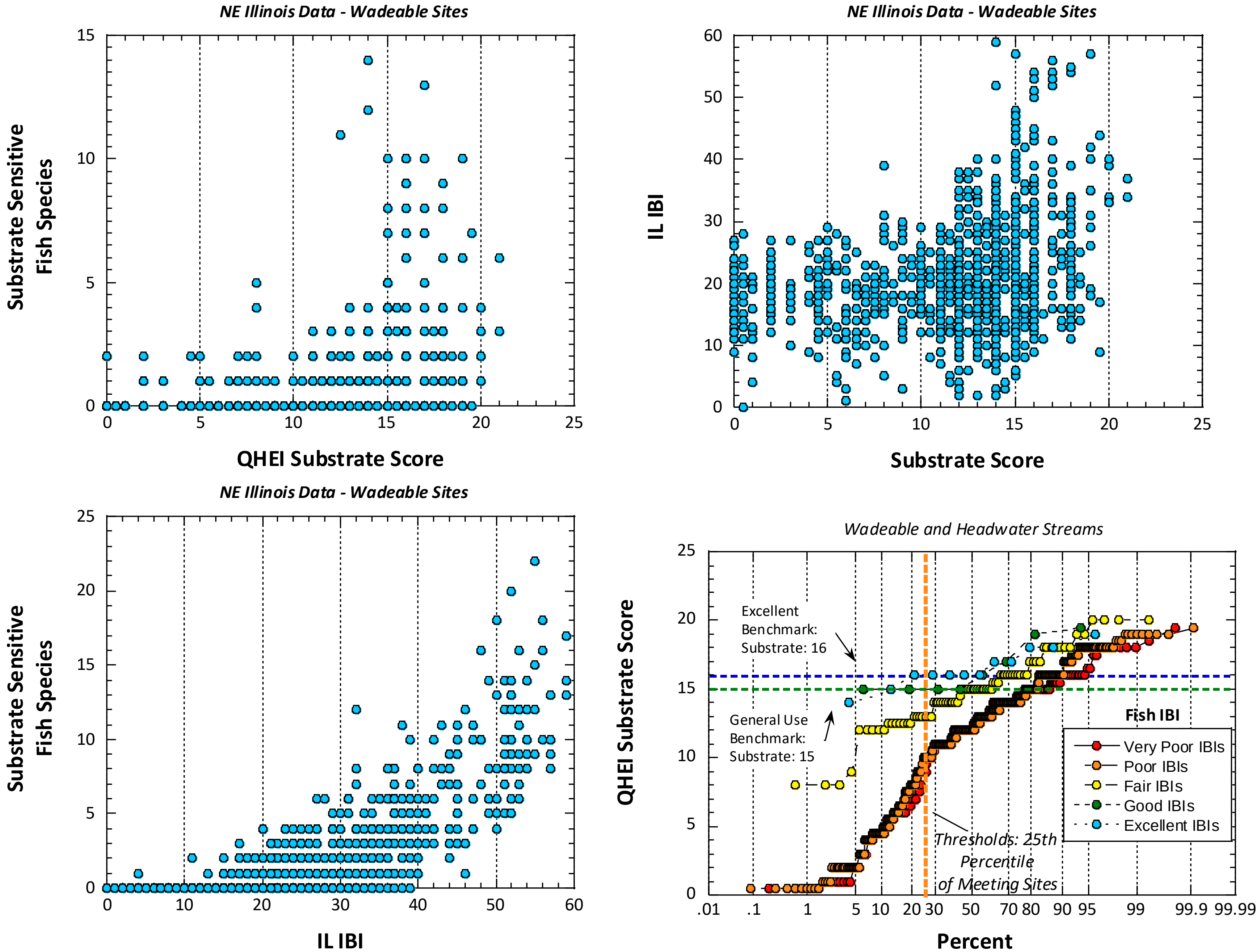
Appendix B includes plots of the data used to derive the IPS effect thresholds for NE Illinois streams and rivers draining <350 $mi.^2$ for chemical, habitat, and land use parameters that showed a relationship with the biological response measured by parameter sensitive fish species and macroinvertebrate taxa. For each parameter there are four plots for the most sensitive of the two aquatic assemblages, fish or macroinvertebrates. These include:

1) A scatter plot of the stressor parameter vs. the number of stressor-sensitive fish species or macroinvertebrate taxa (upper left);
2) A scatter plot of the fish IBI (fIBI) or macroinvertebrate IBI (mIBI) vs. the number of stressor-specific fish species or macroinvertebrate taxa (lower left);
3) A scatter plot of the stressor values vs. the fIBI or mIBI (upper right); and,
4) A probability plot of the stressor by fish IBI or mIBI range with sites in the good and excellent narrative range excluding sites with <$25^{th}$ percentile value of stressor-specific fish species or macroinvertebrate taxa.

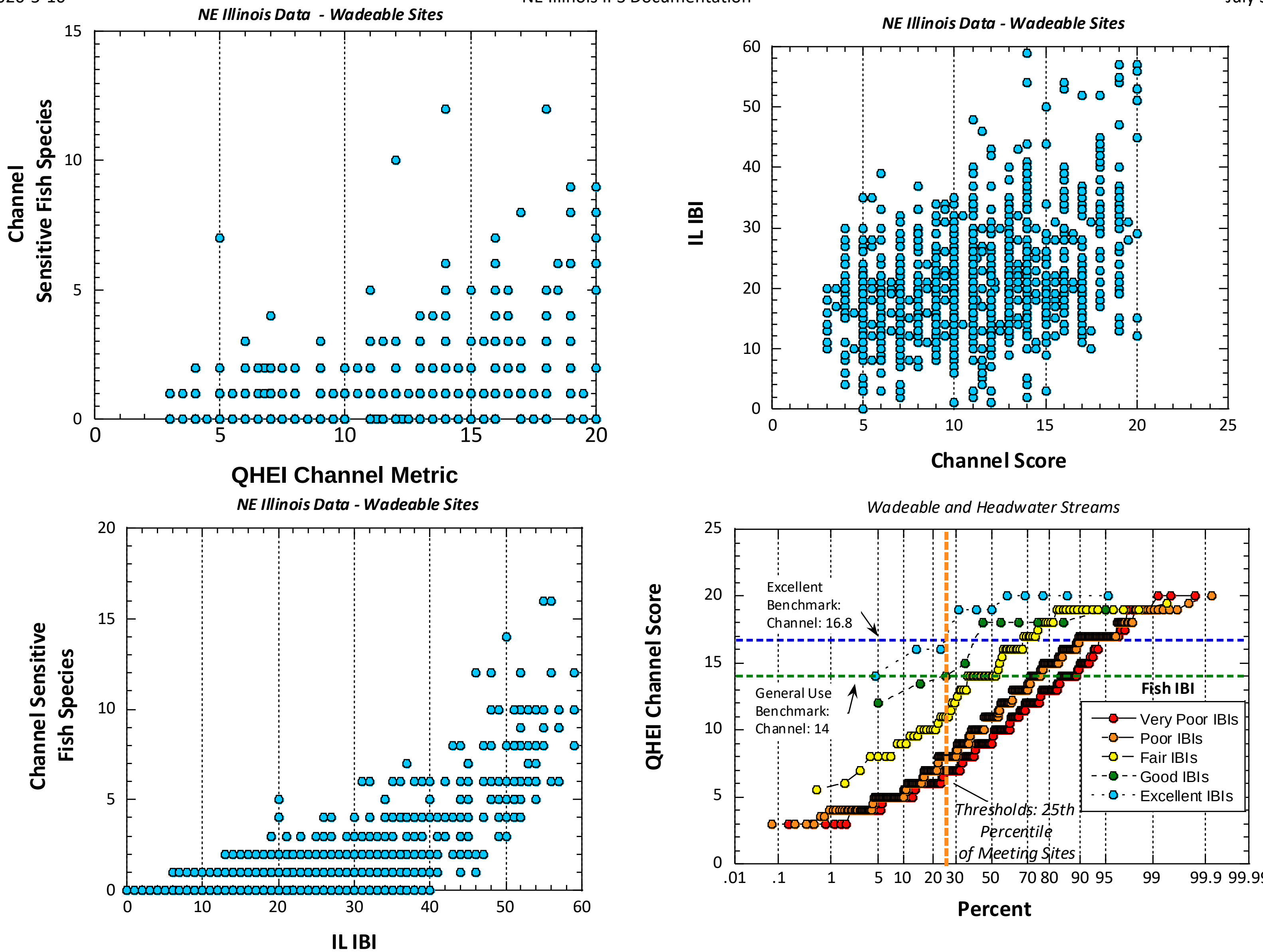
The $75^{th}$ percentile of these distributions of the most sensitive of the fish or macroinvertebrate results was used to set the effect thresholds for that stressor for the Excellent narrative category (Excellent fIBI or mIBI benchmark), the Good narrative (Good fIBI or mIBI benchmark) that is equivalent to the General Use for aquatic life, and the Fair, Poor, and Very Poor narratives respectively.



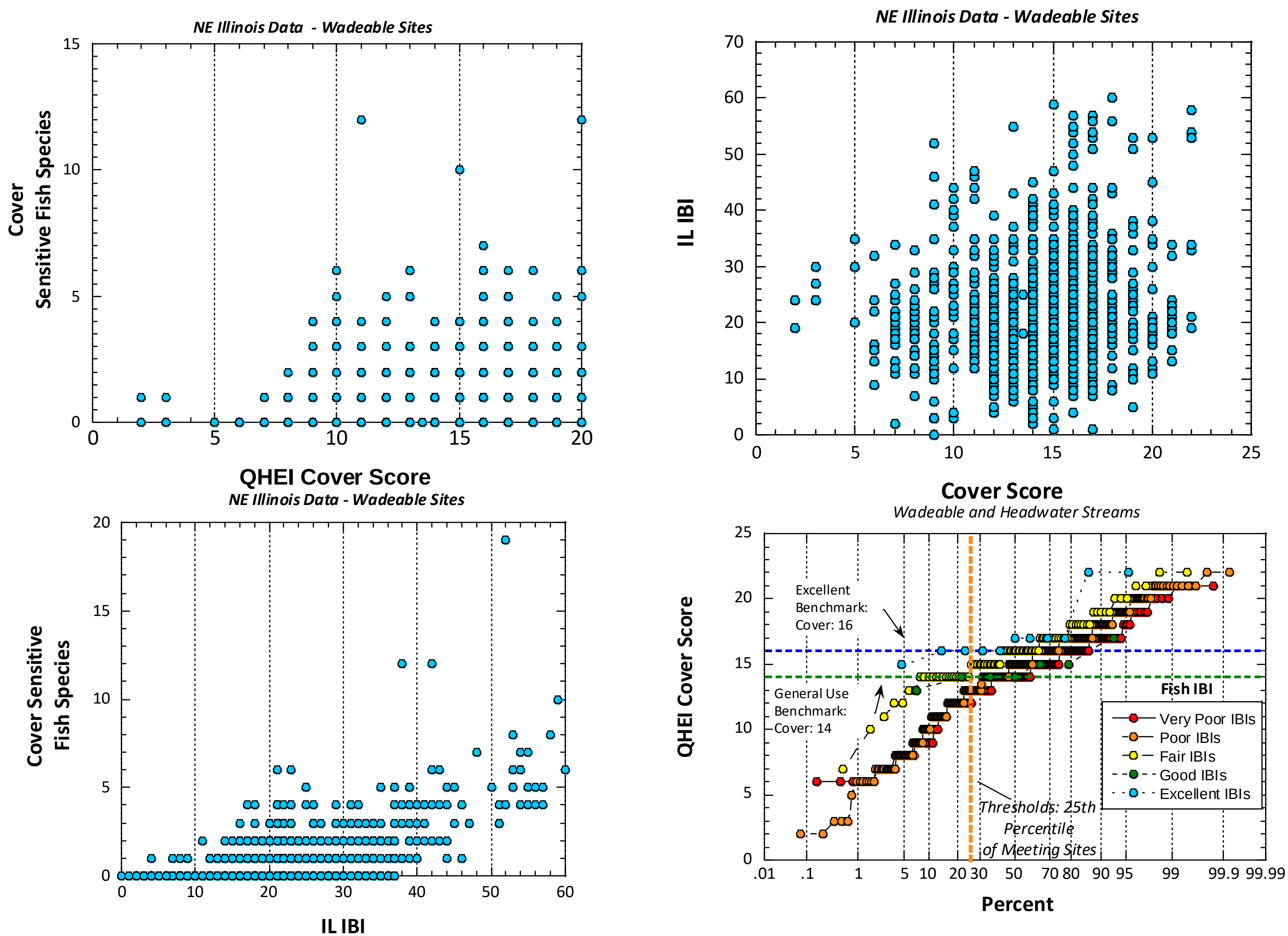

***Appendix Figure B-1.*** *Plots supporting derivation of QHEI total score thresholds for wadeable streams in NE Illinois including scatter plots of QHEI vs QHEI sensitive fish species (top left), fish IBI vs. QHEI sensitive fish species (bottom left), QHEI vs. fish IBI (top right) and a probability plot of QHEI values by narrative ranges of the IBI. Data from IL IPS study area sites in NE Illinois (see text).*



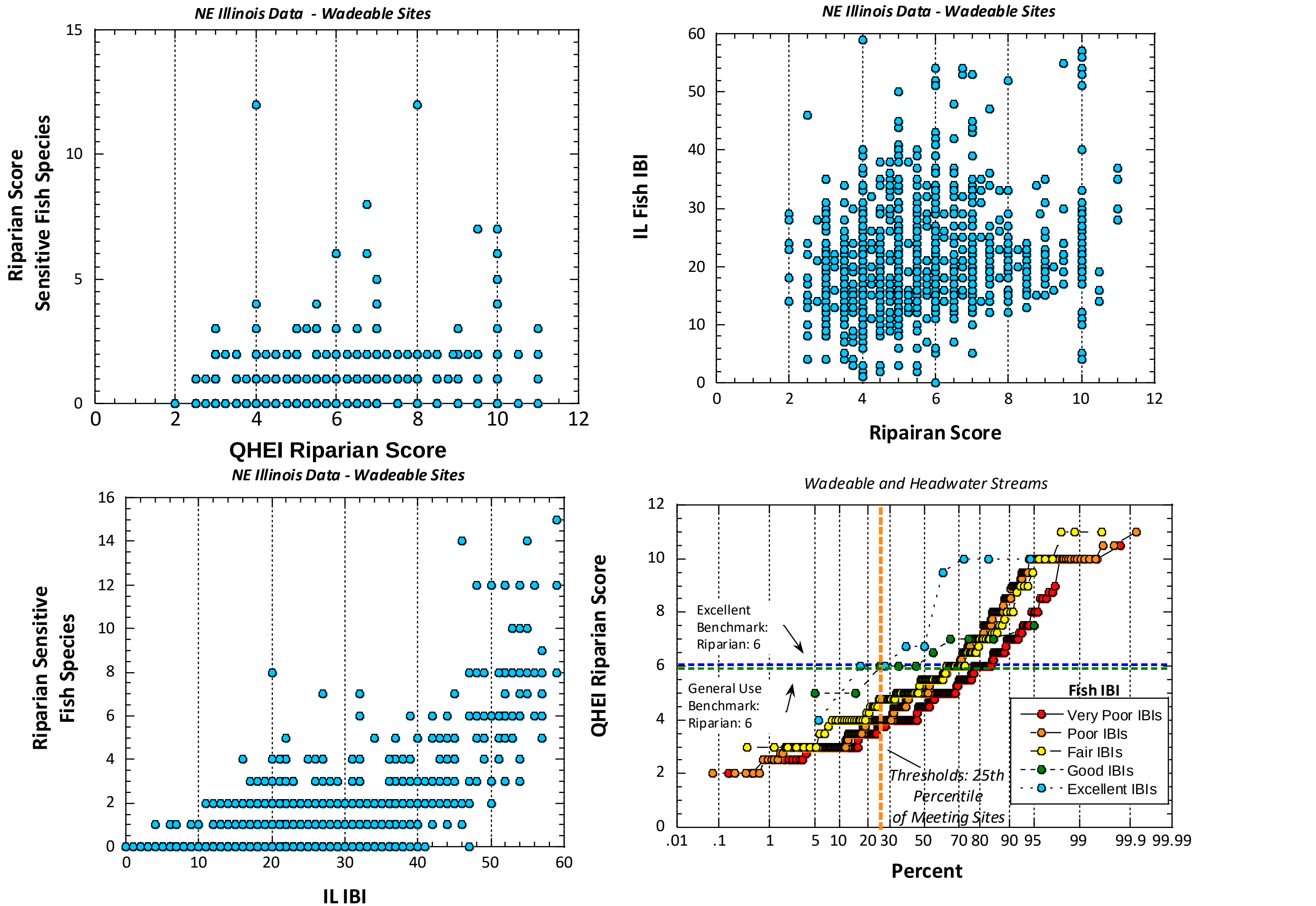

***Appendix Figure B-2.*** *Plots supporting derivation of QHEI substrate score thresholds for wadeable streams in NE Illinois including scatter plots of substrate vs substrate sensitive fish species (top left), fish IBI vs. substrate sensitive fish species (bottom left), substrate vs. fish IBI (top right) and a probability plot of substrate values by narrative ranges of the IBI. Data from IL IPS study area sites in NE Illinois (see text).*



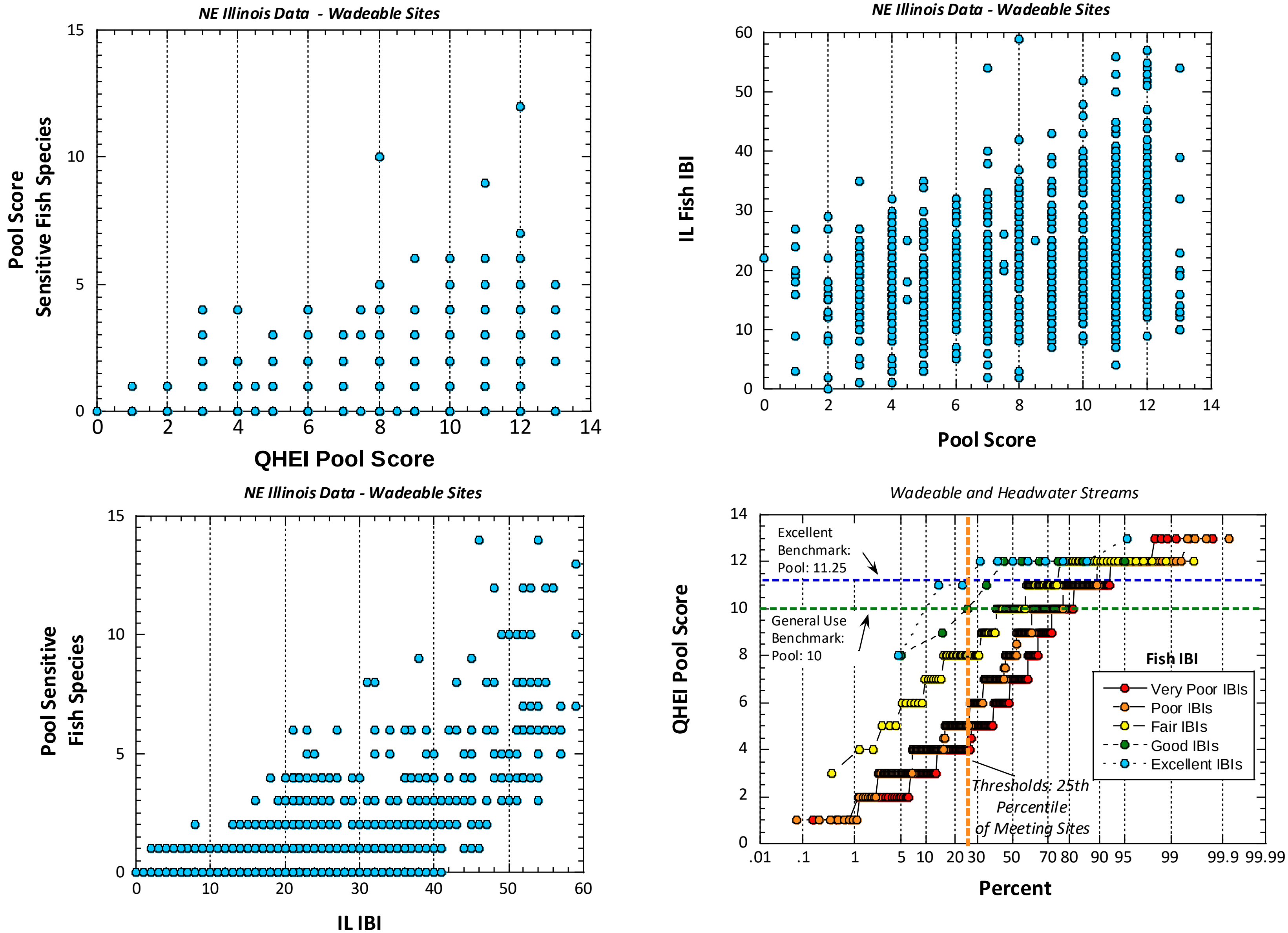

***Appendix Figure B-3***. *Plots supporting derivation of QHEI channel score benchmarks for wadeable streams in NE Illinois including scatter plots of channel vs channel sensitive fish species (top left), fish IBI vs. channel score fish species (bottom left), channel vs. fish IBI (top right) and a probability plot of channel values by narrative ranges of the IBI. Data from IL IPS study area sites in NE Illinois (see text).*



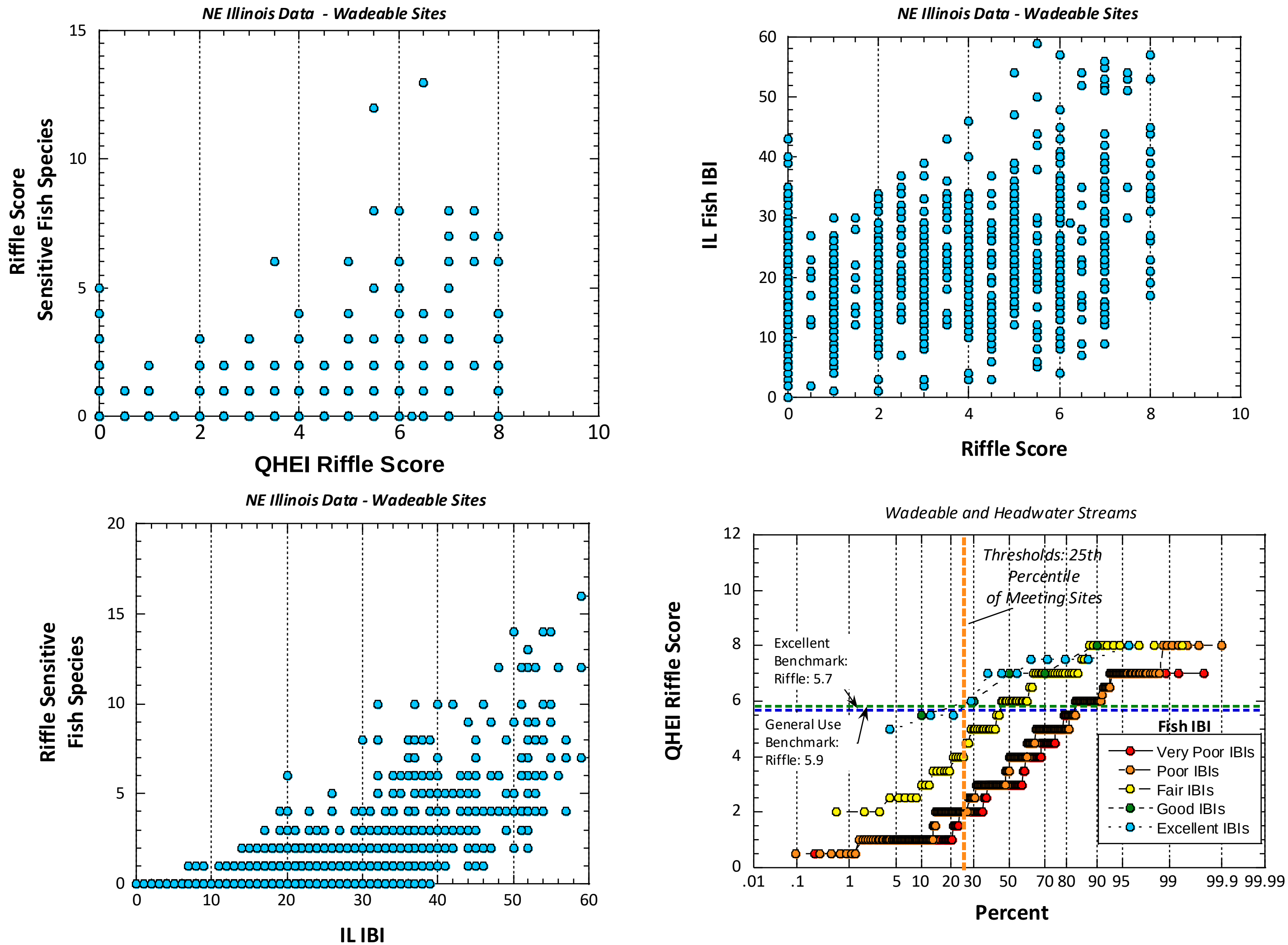

***Appendix Figure B-4****. Plots supporting derivation of QHEI cover score benchmarks for wadeable streams in NE Illinois including scatter plots of cover vs cover sensitive fish species (top left), fish IBI vs. cover sensitive fish species (bottom left), cover vs. fish IBI (top right) and a probability plot of cover values by narrative ranges of the IBI. Data from IL IPS study area sites in NE Illinois (see text).*



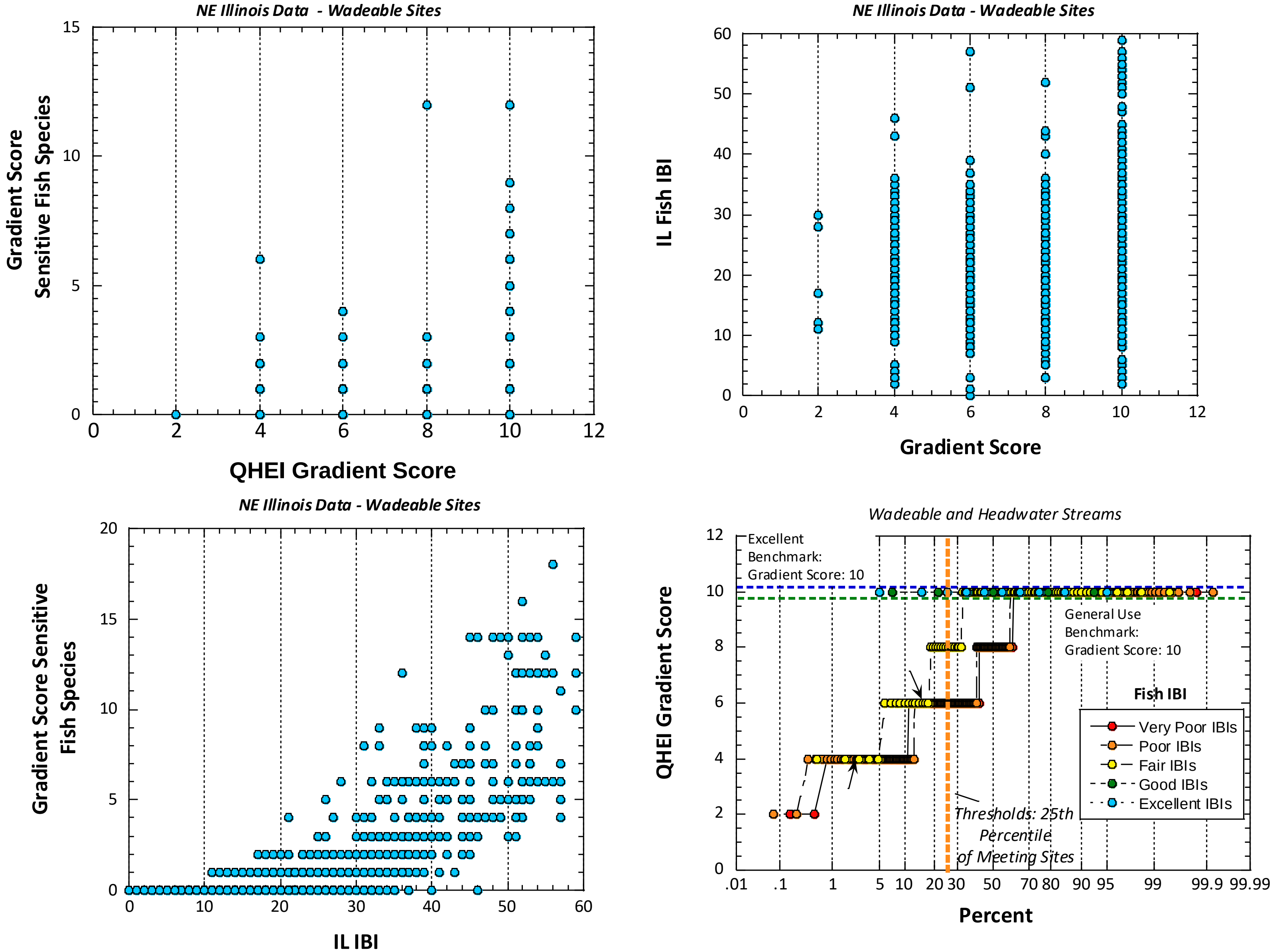

***Appendix Figure B-5.*** *Plots supporting derivation of QHEI riparian score benchmarks for wadeable streams in NE Illinois including scatter plots of riparian vs riparian sensitive fish species (top left), fish IBI vs. riparian sensitive fish species (bottom left), riparian vs. fish IBI (top right) and a probability plot of riparian values by narrative ranges of the IBI. Data from IL IPS study area sites in NE Illinois (see text).*



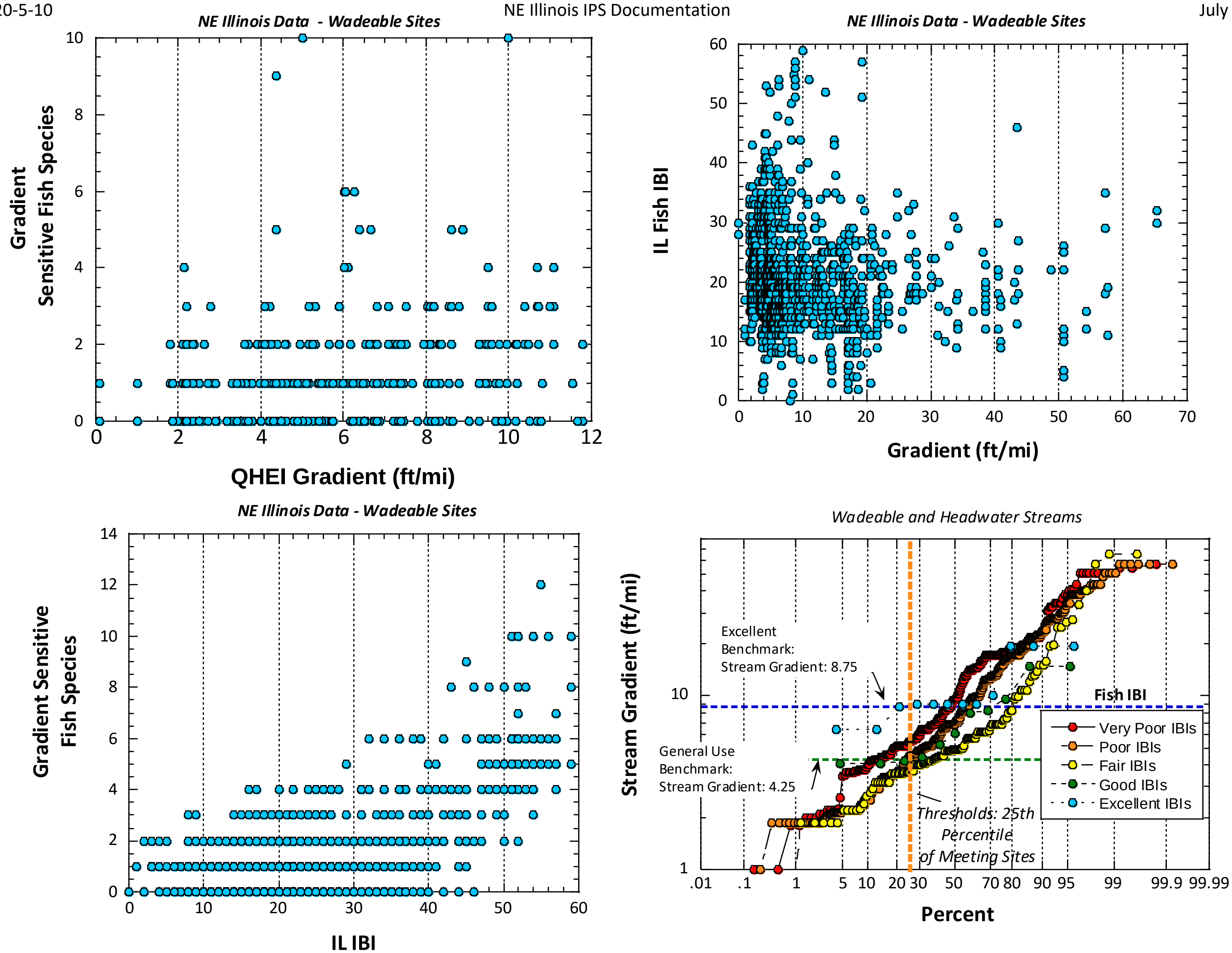

***Appendix Figure B-6.*** *Plots supporting derivation of QHEI pool score benchmarks for wadeable streams in NE Illinois including scatter plots of pool score vs pool sensitive fish species (top left), fish IBI vs. pool sensitive fish species (bottom left), pool score vs. fish IBI (top right) and a probability plot of pool scores by narrative ranges of the IBI. Data from IL IPS study area sites in NE Illinois (see text).*



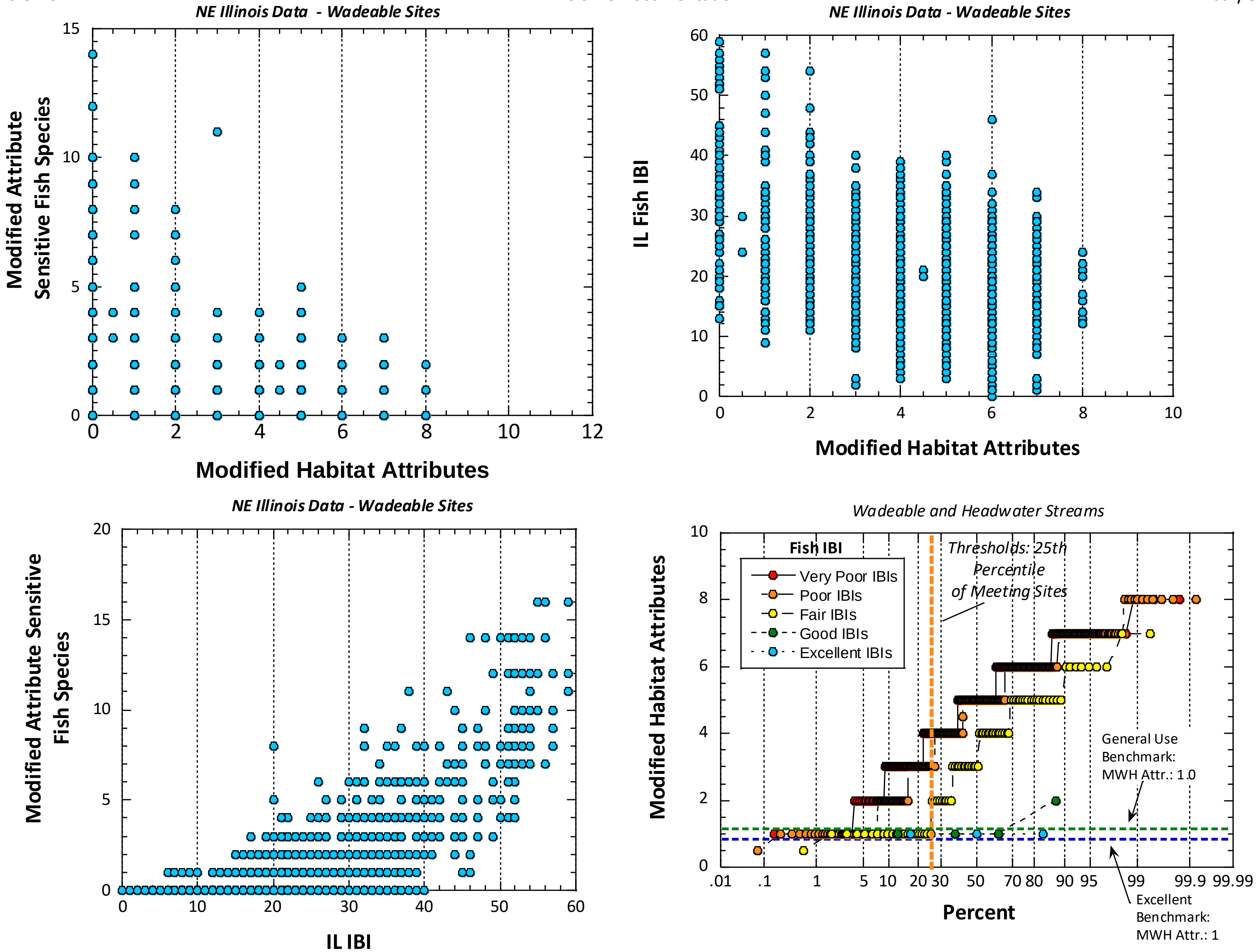

***Appendix Figure B-7.*** *Plots supporting derivation of QHEI riffle score benchmarks for wadeable streams in NE Illinois including scatter plots of riffle score vs riffle sensitive fish species (top left), fish IBI vs. riffle sensitive fish species (bottom left), riffle score vs. fish IBI (top right) and a probability plot of riffle scores by narrative ranges of the IBI. Data from IL IPS study area sites in NE Illinois (see text).*



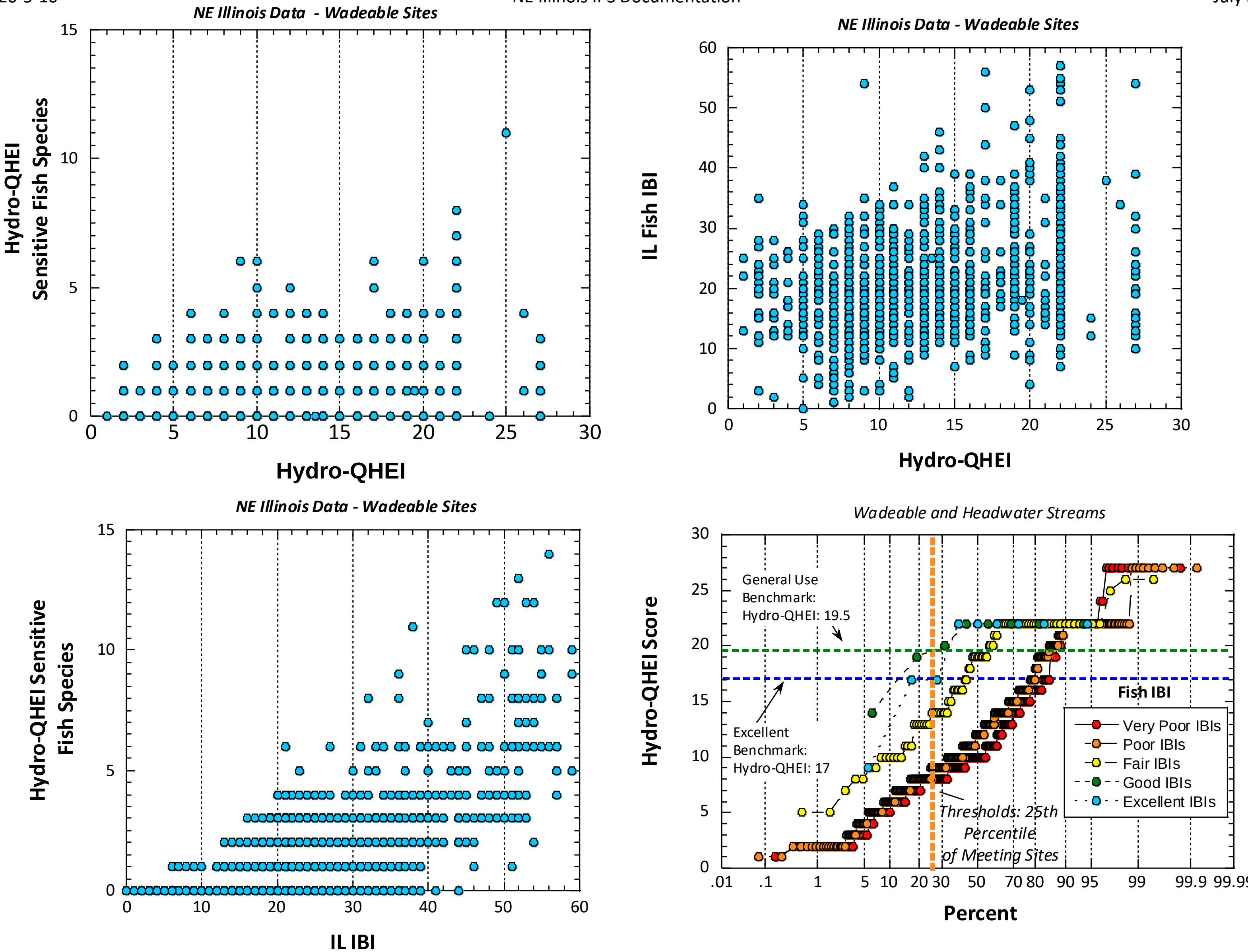

***Appendix Figure B-8.*** *Plots supporting derivation of QHEI gradient score benchmarks for wadeable streams in NE Illinois including scatter plots of gradient score vs gradient sensitive fish species (top left), fish IBI vs. gradient sensitive fish species (bottom left), gradient score vs. fish IBI (top right) and a probability plot of gradient scores by narrative ranges of the IBI. Data from IL IPS study area sites in NE Illinois (see text).*



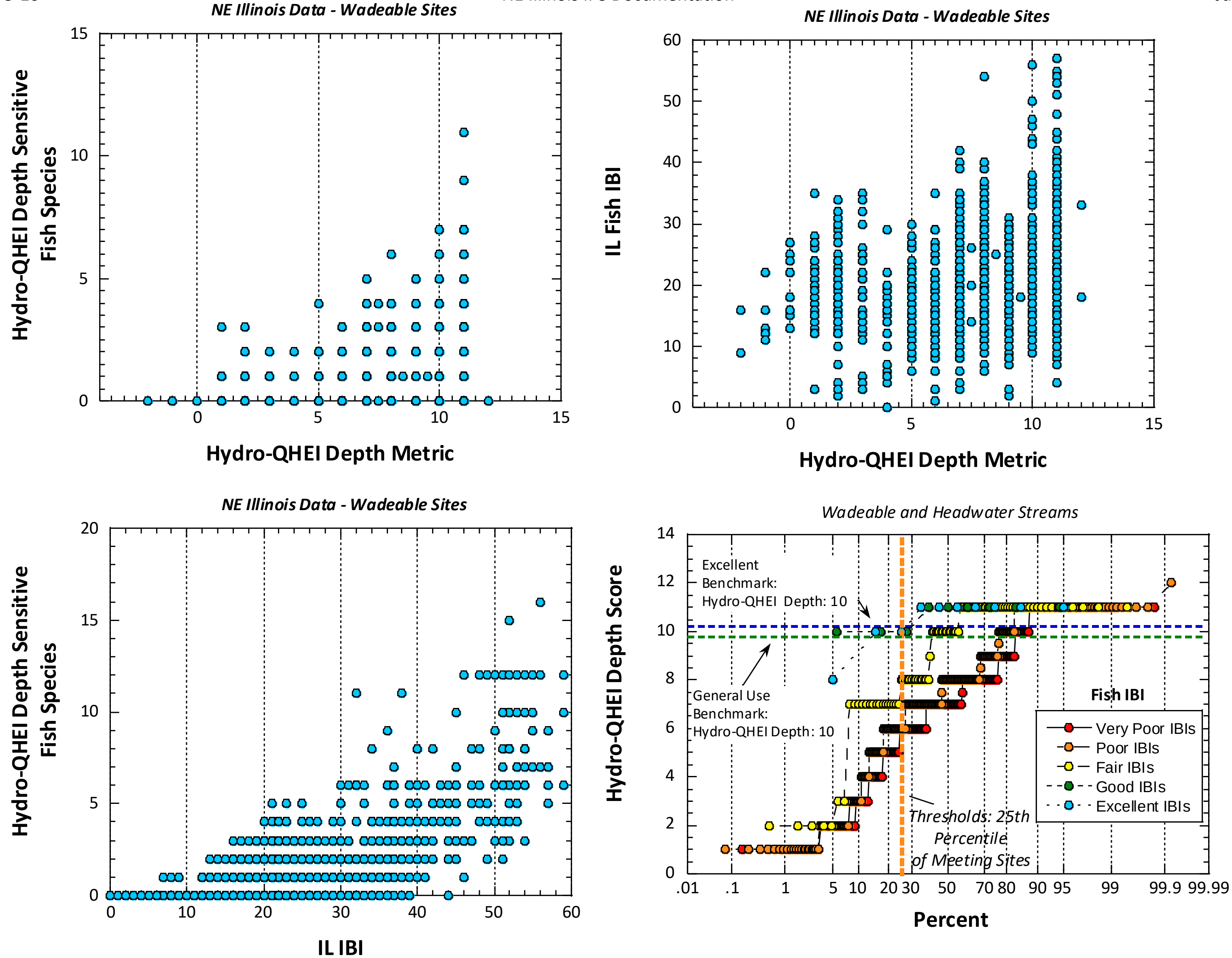

***Appendix Figure B-9.*** *Plots supporting derivation of QHEI gradient benchmarks for wadeable streams in NE Illinois including scatter plots of gradient vs gradient sensitive fish species (top left), fish IBI vs. gradient sensitive fish species (bottom left), gradient vs. fish IBI (top right) and a probability plot of gradient by narrative ranges of the IBI. Data from IL IPS study area sites in NE Illinois (see text).*



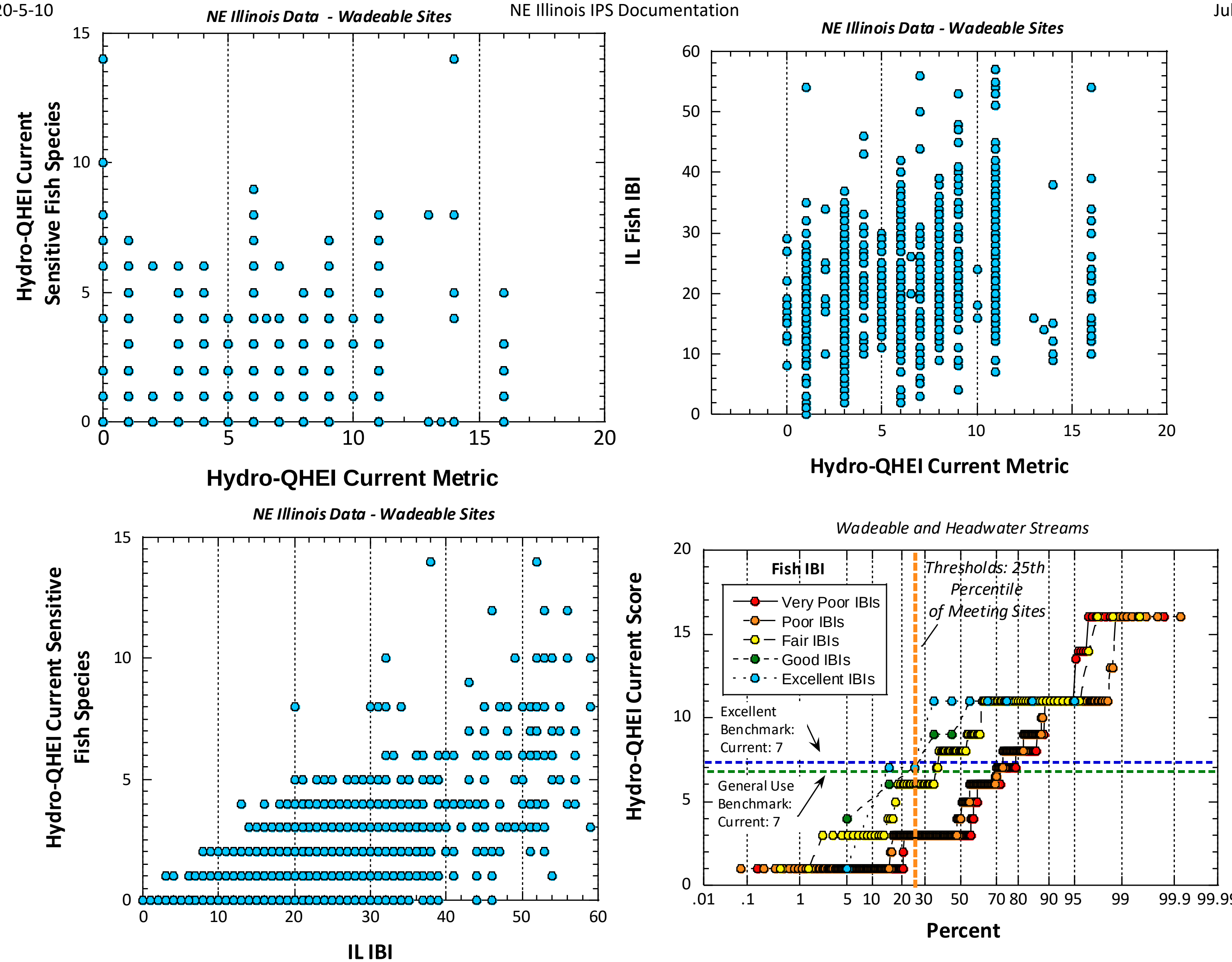

***Appendix Figure B-10.*** *Plots supporting derivation of QHEI gradient benchmarks for wadeable streams in NE Illinois including scatter plots of poor attributes vs poor attribute sensitive fish species (top left), fish IBI vs. poor attribute sensitive fish species (bottom left), poor attribute vs. fish IBI (top right) and a probability plot of poor habitat attributes by narrative ranges of the IBI. Data from IL IPS study area sites in NE Illinois (see text).*



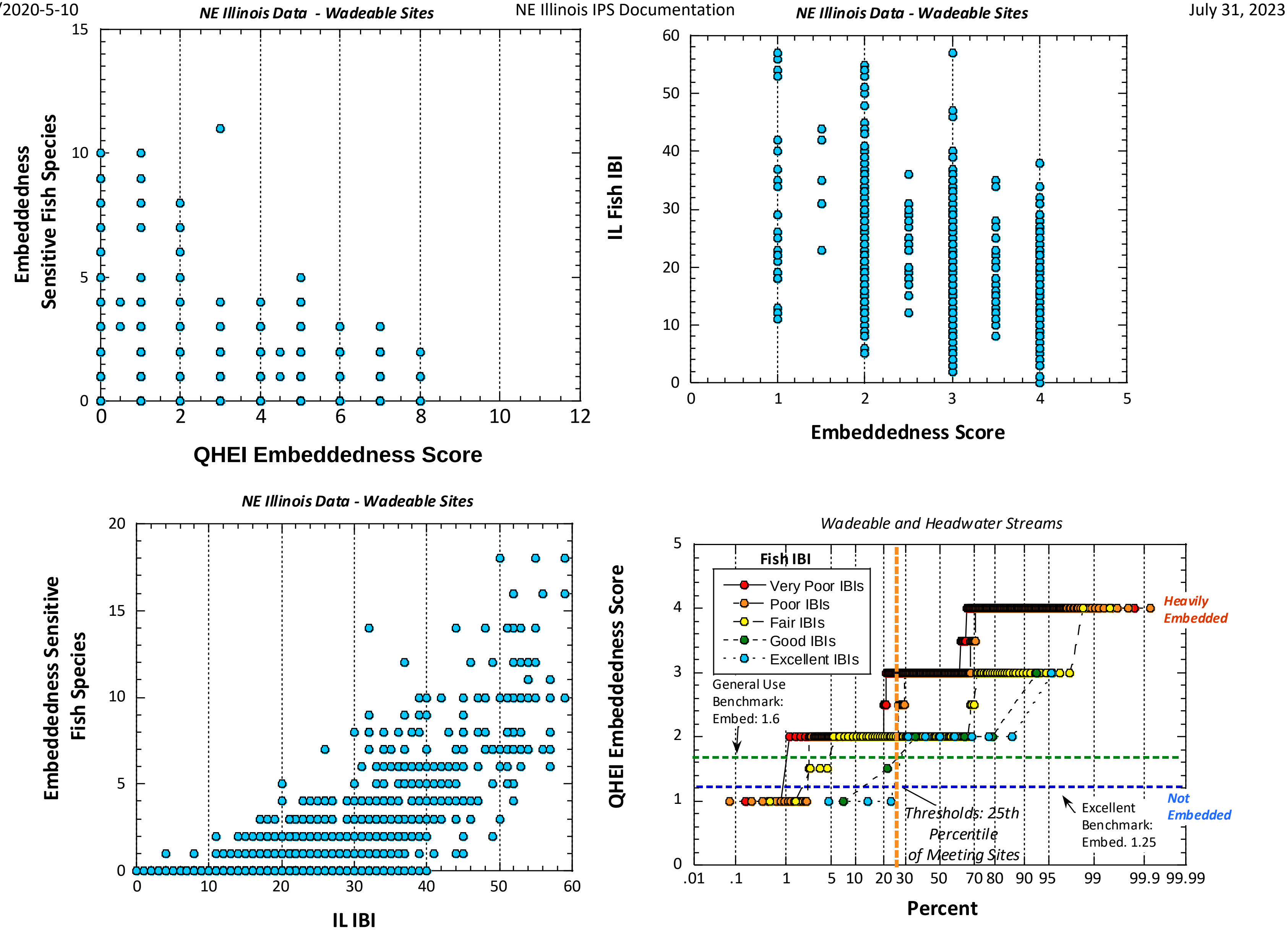

***Appendix Figure B-11.*** *Plots supporting derivation of QHEI Hydro-QHEI benchmarks for wadeable streams in NE Illinois including scatter plots of Hydro-QHEI vs Hydro-QHEI sensitive fish species (top left), fish IBI vs. Hydro-QHEI sensitive fish species (bottom left), Hydro-QHEI vs. fish IBI (top right) and a probability plot of Hydro-QHEI by narrative ranges of the IBI. Data from IL IPS study area sites in NE Illinois (see text).*



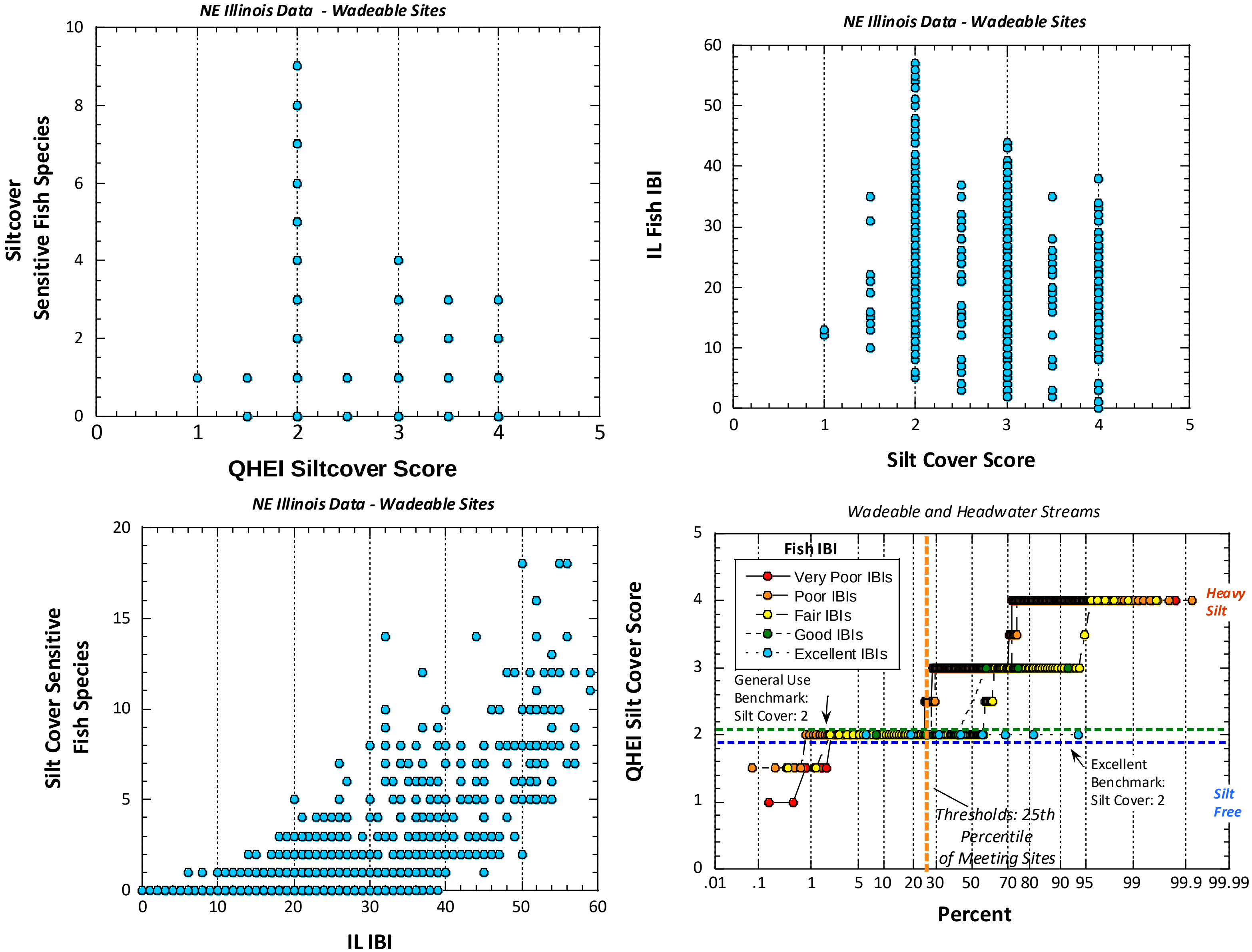

***Appendix Figure B-12.*** *Plots supporting derivation of QHEI Hydro-QHEI depth metric benchmarks for wadeable streams in NE Illinois including scatter plots of Hydro-QHEI depth metric vs Hydro-QHEI depth sensitive fish species (top left), fish IBI vs. Hydro-QHEI depth sensitive fish species (bottom left), Hydro-QHEI depth metric vs. fish IBI (top right) and a probability plot of Hydro-QHEI depth metric by narrative ranges of the IBI. Data from IL IPS study area sites in NE Illinois (see text).*



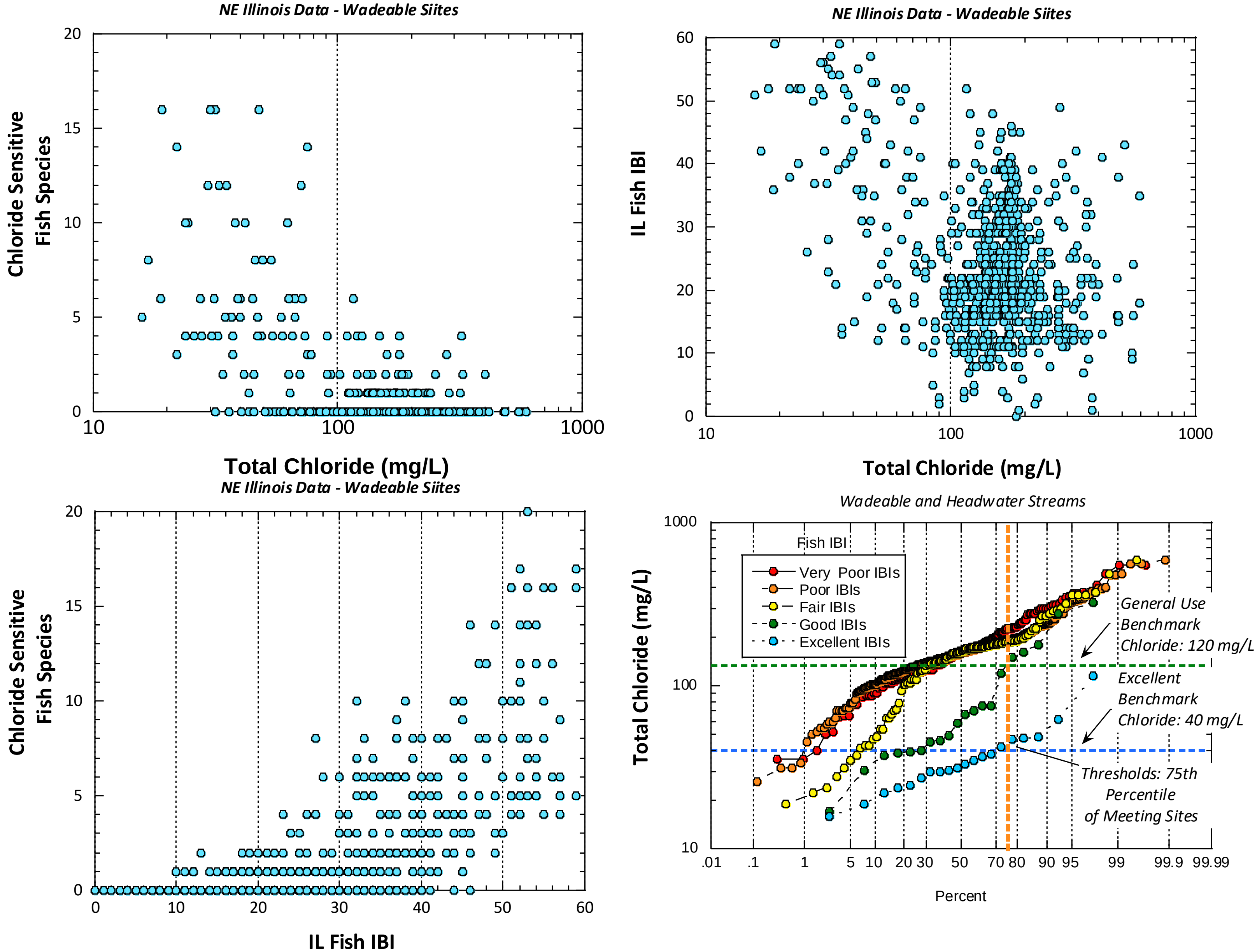

***Appendix Figure B-13.*** *Plots supporting derivation of QHEI Hydro-QHEI current metric benchmarks for wadeable streams in NE Illinois including scatter plots of Hydro-QHEI current metric vs Hydro-QHEI current sensitive fish species (top left), fish IBI vs. Hydro-QHEI current sensitive fish species (bottom left), Hydro-QHEI current metric vs. fish IBI (top right) and a probability plot of Hydro-QHEI current metric by narrative ranges of the IBI. Data from IL IPS study area sites in NE Illinois (see text).*



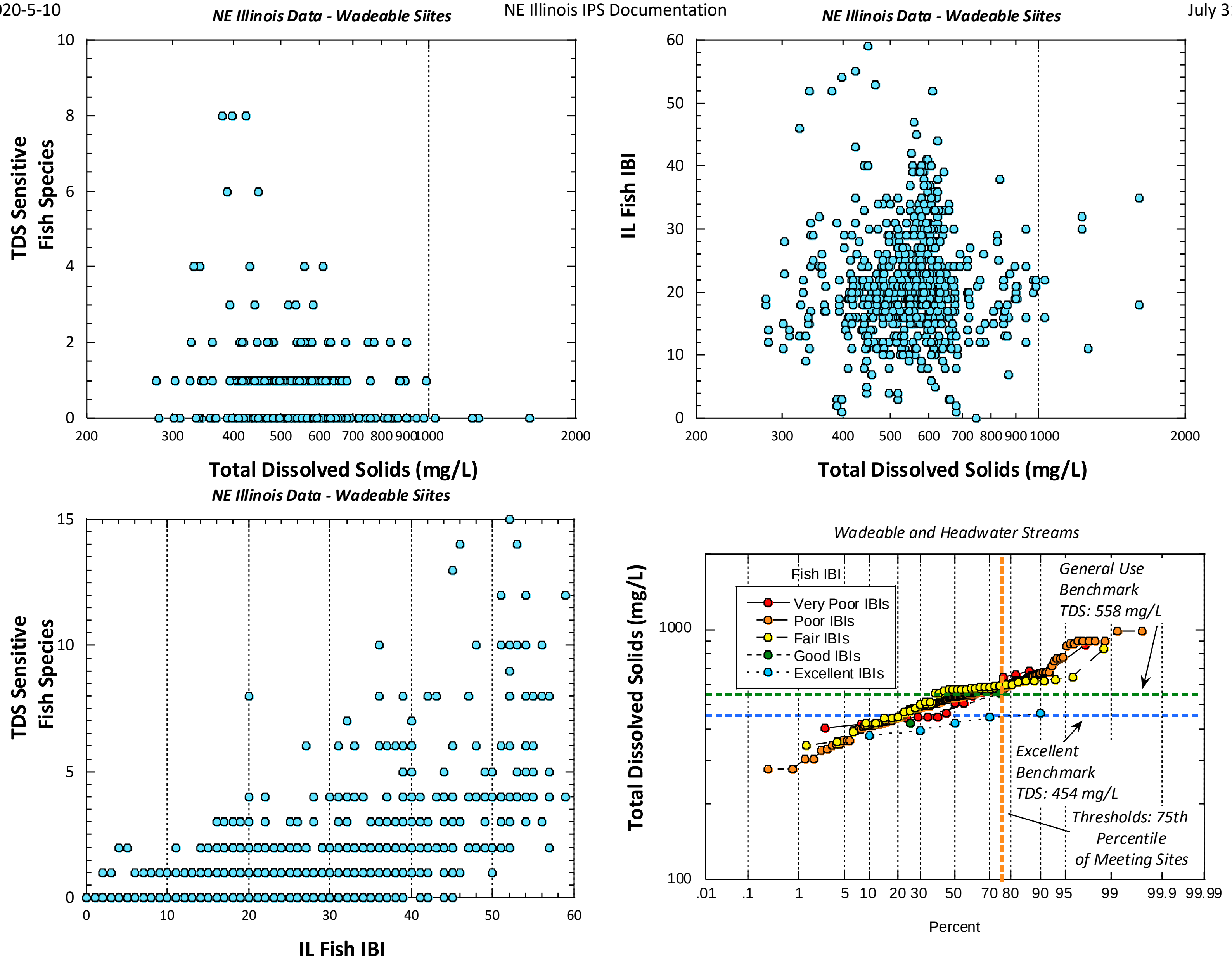

***Appendix Figure B-14.*** *Plots supporting derivation of QHEI embeddedness score benchmarks for wadeable streams in NE Illinois including scatter plots of embeddedness score vs embeddedness sensitive fish species (top left), fish IBI vs. embeddedness sensitive fish species (bottom left), embeddedness score vs. fish IBI (top right) and a probability plot of embeddedness score by narrative ranges of the IBI. Data from IL IPS study area sites in NE Illinois (see text).*



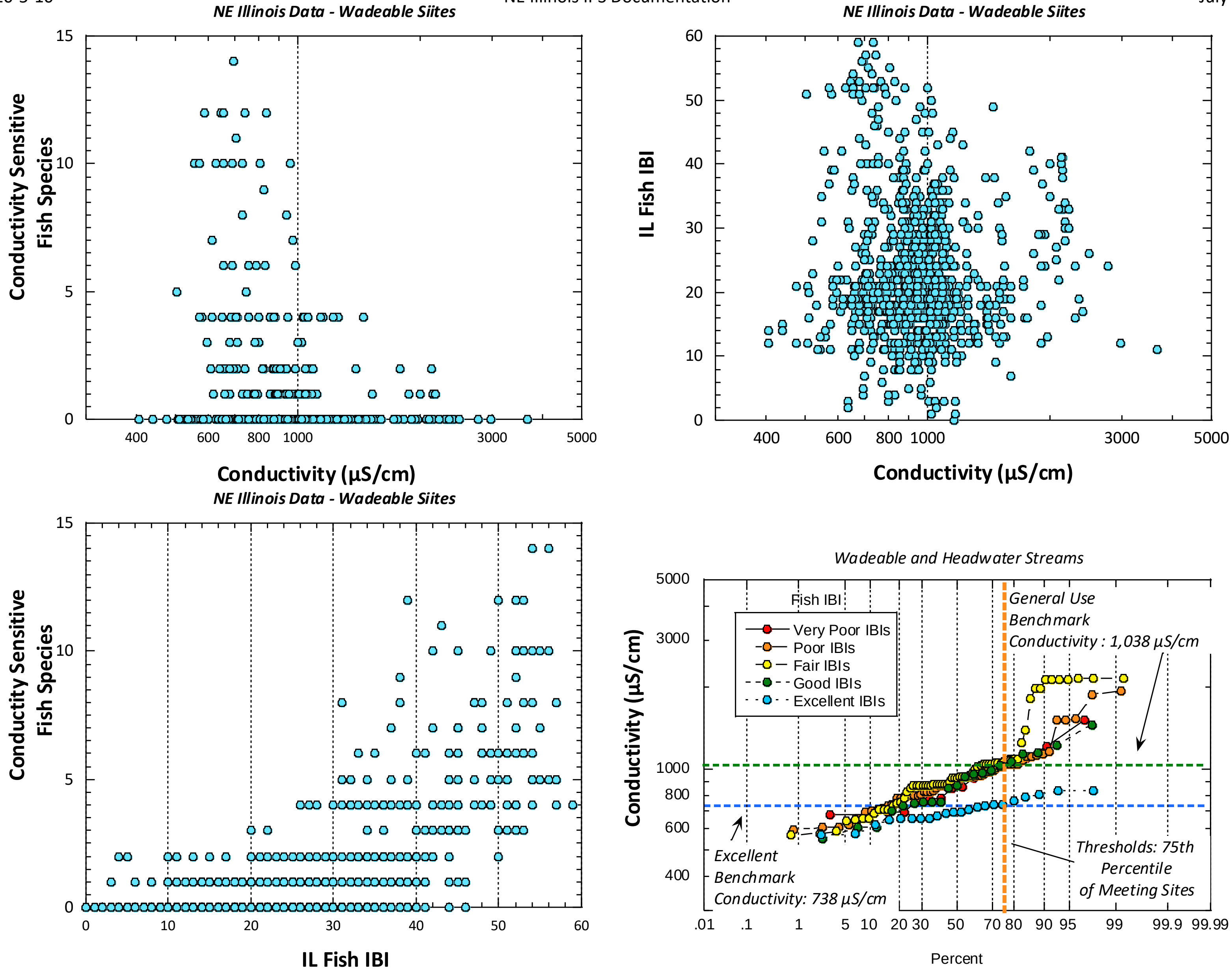

***Appendix Figure B-15.*** *Plots supporting derivation of QHEI silt cover score benchmarks for wadeable streams in NE Illinois including scatter plots of silt cover score vs silt cover sensitive fish species (top left), fish IBI vs. silt cover sensitive fish species (bottom left), silt cover score vs. fish IBI (top right) and a probability plot of silt cover score by narrative ranges of the IBI. Data from IL IPS study area sites in NE Illinois (see text).*



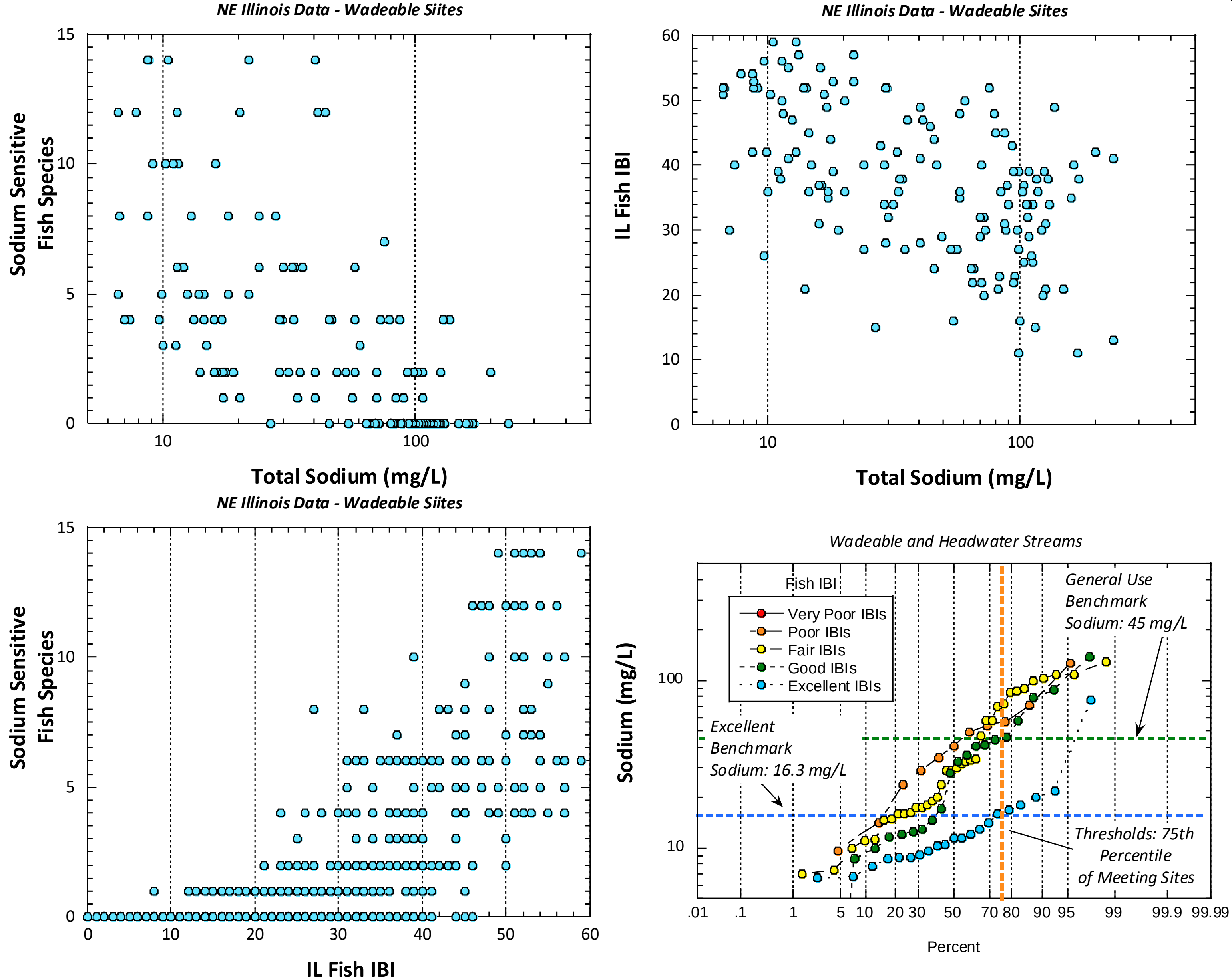

***Appendix Figure B-16.*** *Plots supporting the derivation of total chloride thresholds for NE Illinois streams and rivers draining <350 mi.² including scatter plots of total chloride vs chloride sensitive fish species (top left), fIBI vs. chloride sensitive fish species (bottom left), total chloride vs. fIBI (top right), and a probability plot of total chloride by narrative ranges of the fIBI.*



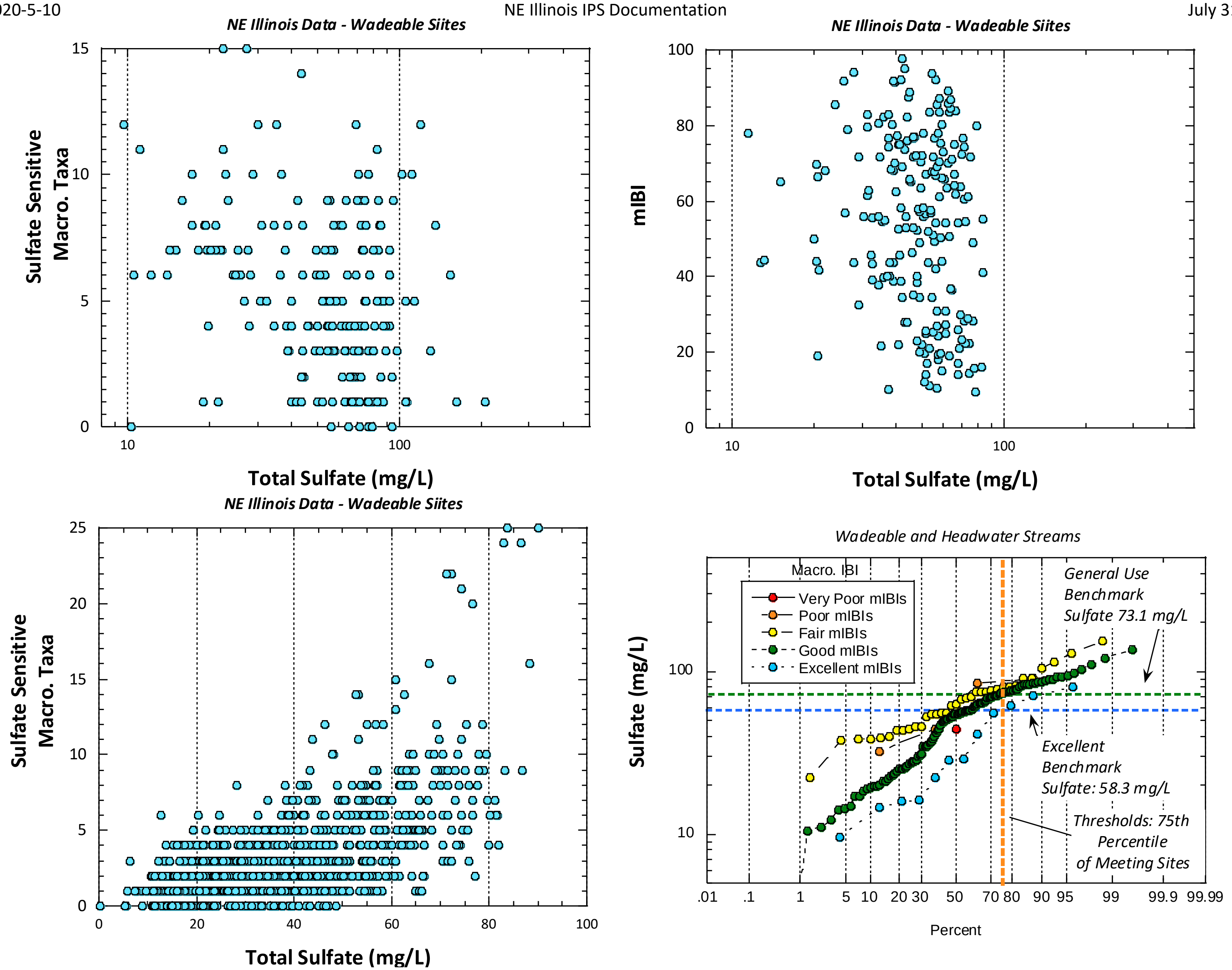

***Appendix Figure B-17****. Plots supporting derivation of total dissolved solids (TDS) thresholds for NE Illinois rivers and streams including scatter plots of TDS vs. TDS sensitive fish species (top left), fIBI vs. TDS sensitive fish species (bottom left), TDS vs. fIBI (top right) and a probability plot of TDS by narrative ranges of the fIBI.*



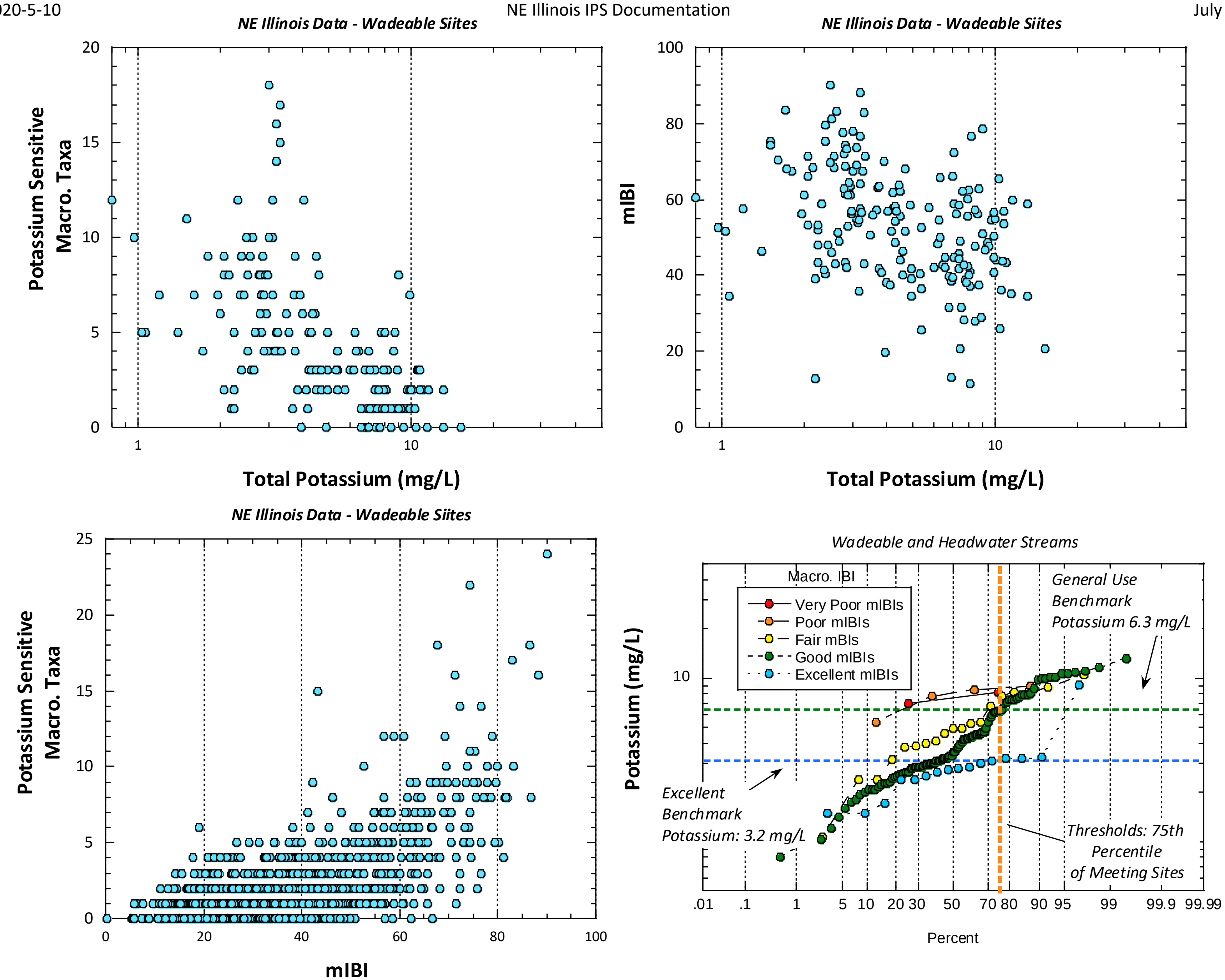

***Appendix Figure B-18.*** *Plots supporting derivation of conductivity thresholds for NE Illinois rivers and streams including scatter plots of conductivity vs conductivity sensitive fish species (top left), fIBI vs. conductivity sensitive fish species (bottom left), conductivity vs. fIBI (top right) and a probability plot of conductivity by narrative ranges of the fIBI.*



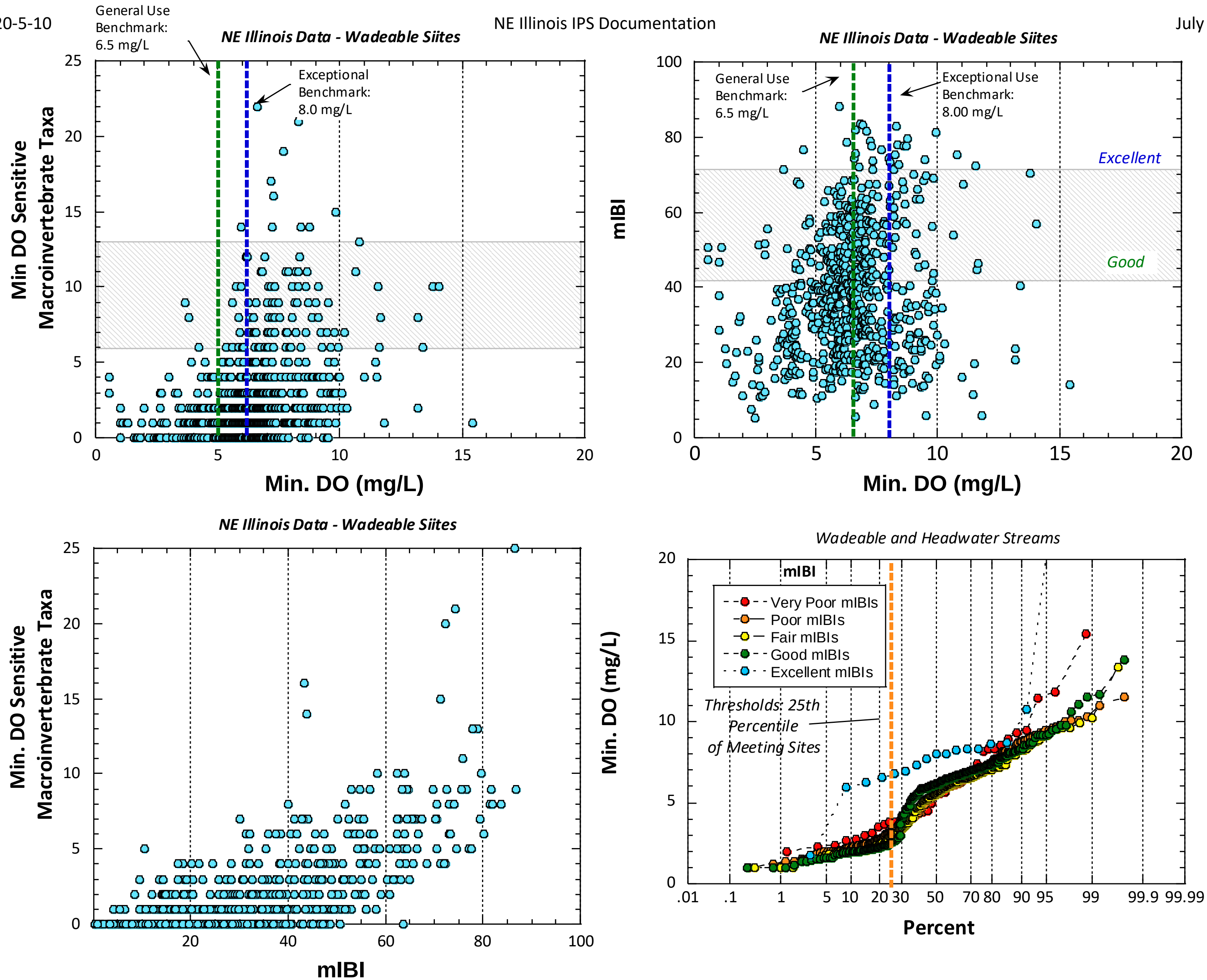

***Appendix Figure B-19.*** *Plots supporting derivation of sodium benchmarks for wadeable streams in NE Illinois including scatter plots of sodium vs sodium sensitive fish species (top left), fish IBI vs. sodium sensitive fish species (bottom left), sodium vs. fish IBI (top right) and a probability plot of sodium by narrative ranges of the IBI. Data from IL IPS study area sites in NE Illinois (see text).*



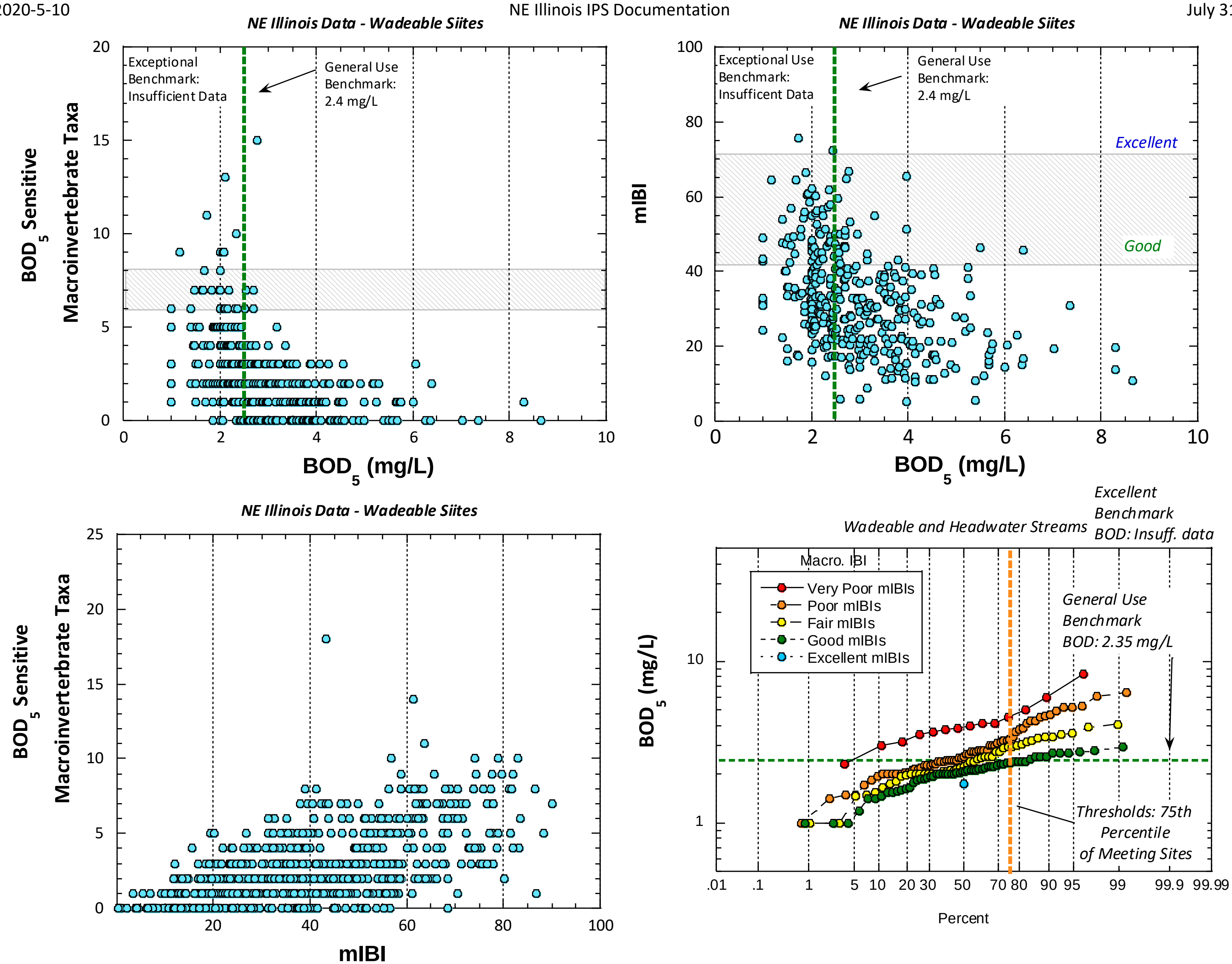

***Appendix Figure B-20.*** *Plots supporting derivation of sulfate thresholds for NE Illinois rivers and streams including scatter plots of sulfate vs sulfate sensitive macroinvertebrate taxa (top left), mIBI vs. sulfate sensitive macroinvertebrate taxa (bottom left), sulfate vs. mIBI (top right) and a probability plot of sulfate by narrative ranges of the mIBI.*



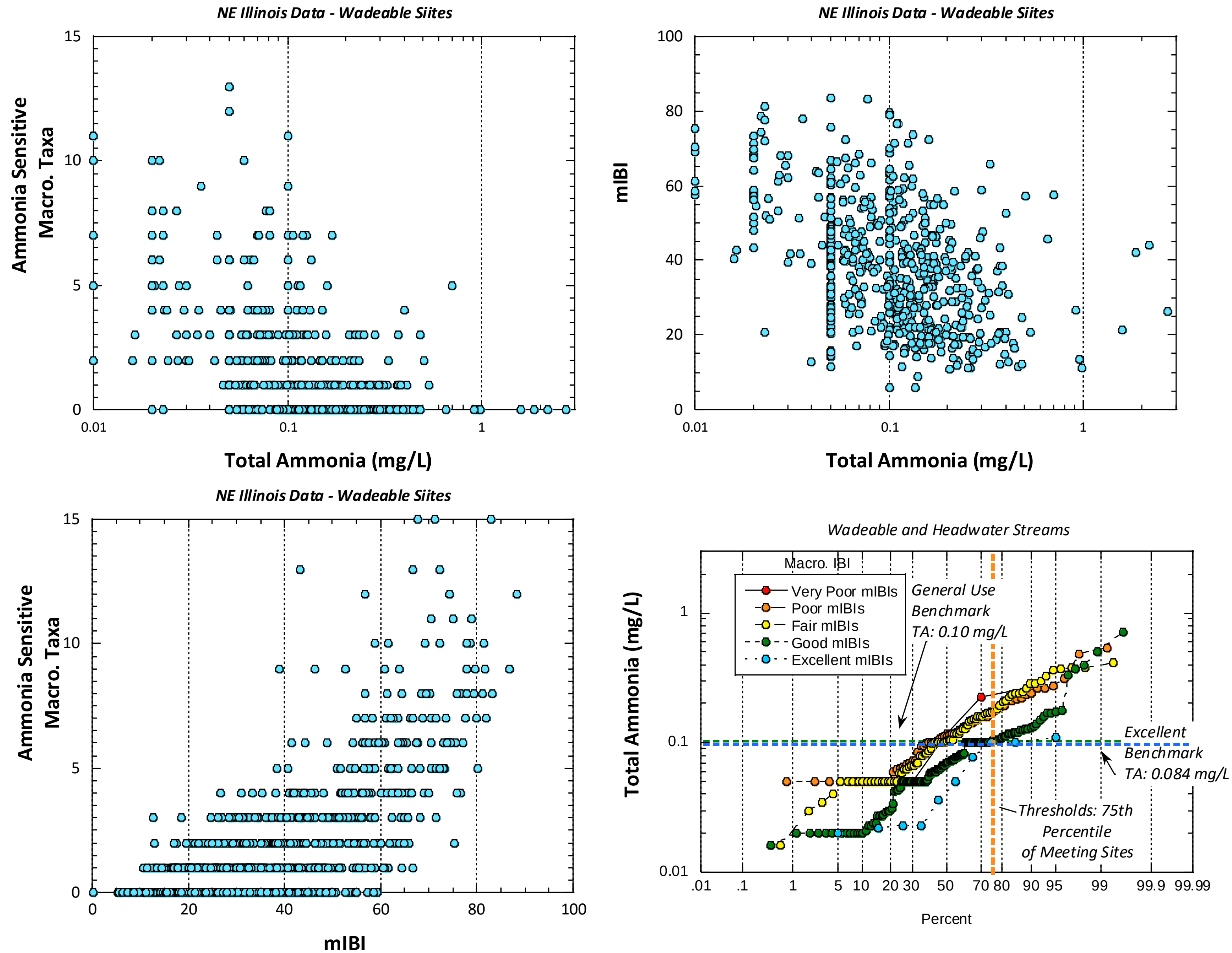

***Appendix Figure B-21.*** *Plots supporting derivation of potassium benchmarks for wadeable streams in NE Illinois including scatter plots of potassium vs potassium sensitive macroinvertebrate taxa (top left), mIBI vs. potassium sensitive macroinvertebrate taxa (bottom left), potassium vs. mIBI (top right) and a probability plot of potassium by narrative ranges of the mIBI. Data from IL IPS study area sites in NE Illinois*



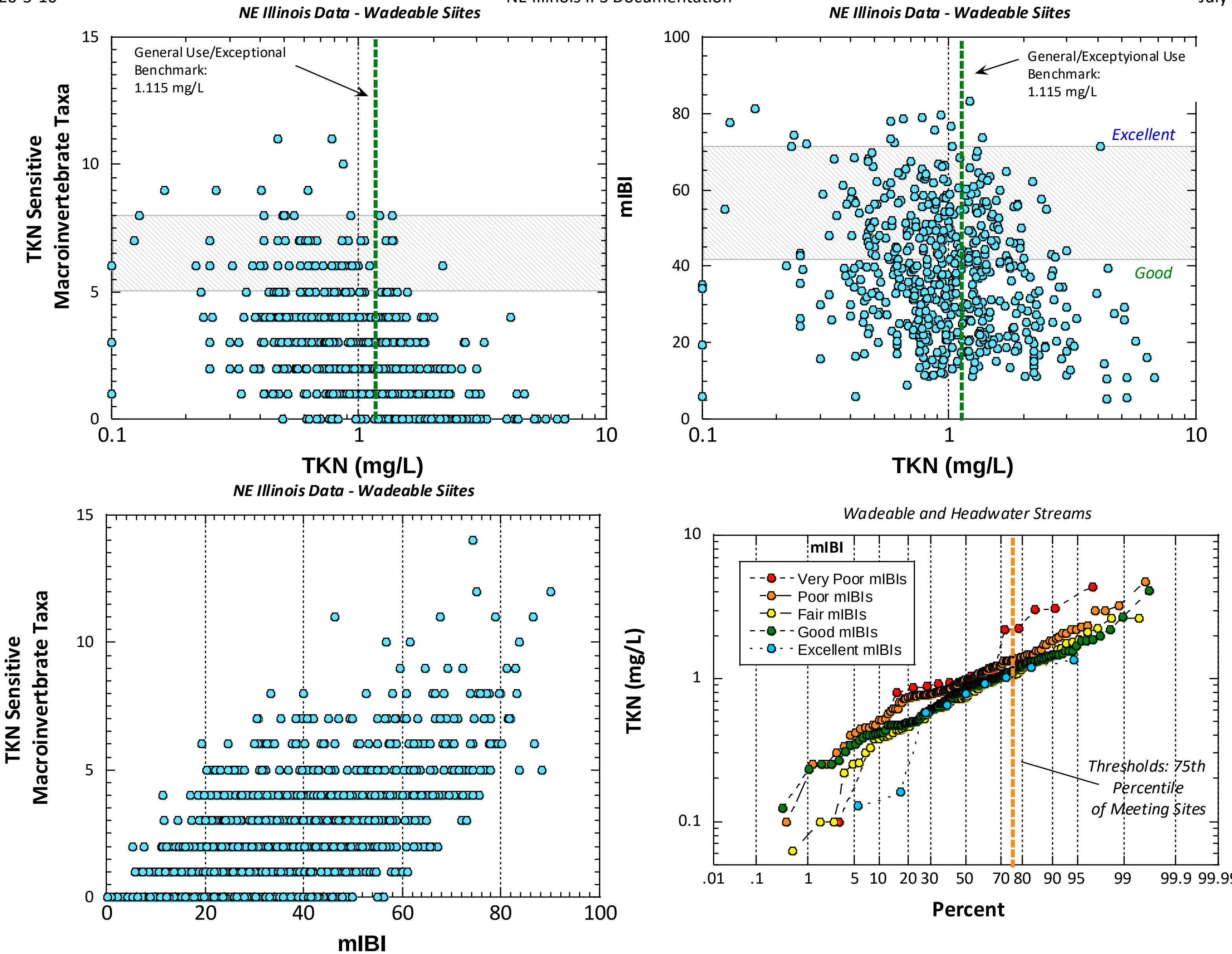

***Appendix Figure B-22***. *Plots supporting derivation to Min. DO benchmarks for wadeable streams in NE Illinois including scatter plots of Min. DO vs Min. DO-macroinvertebrate taxa (top left), IBI vs. Min. DO-sensitive macroinvertebrate taxa (bottom left), Min. DO vs. mIBI (top right) and a probability plot of Min. DO values by narrative ranges of the mIBI. Data from IL IPS study area sites in NE Illinois (see text).*



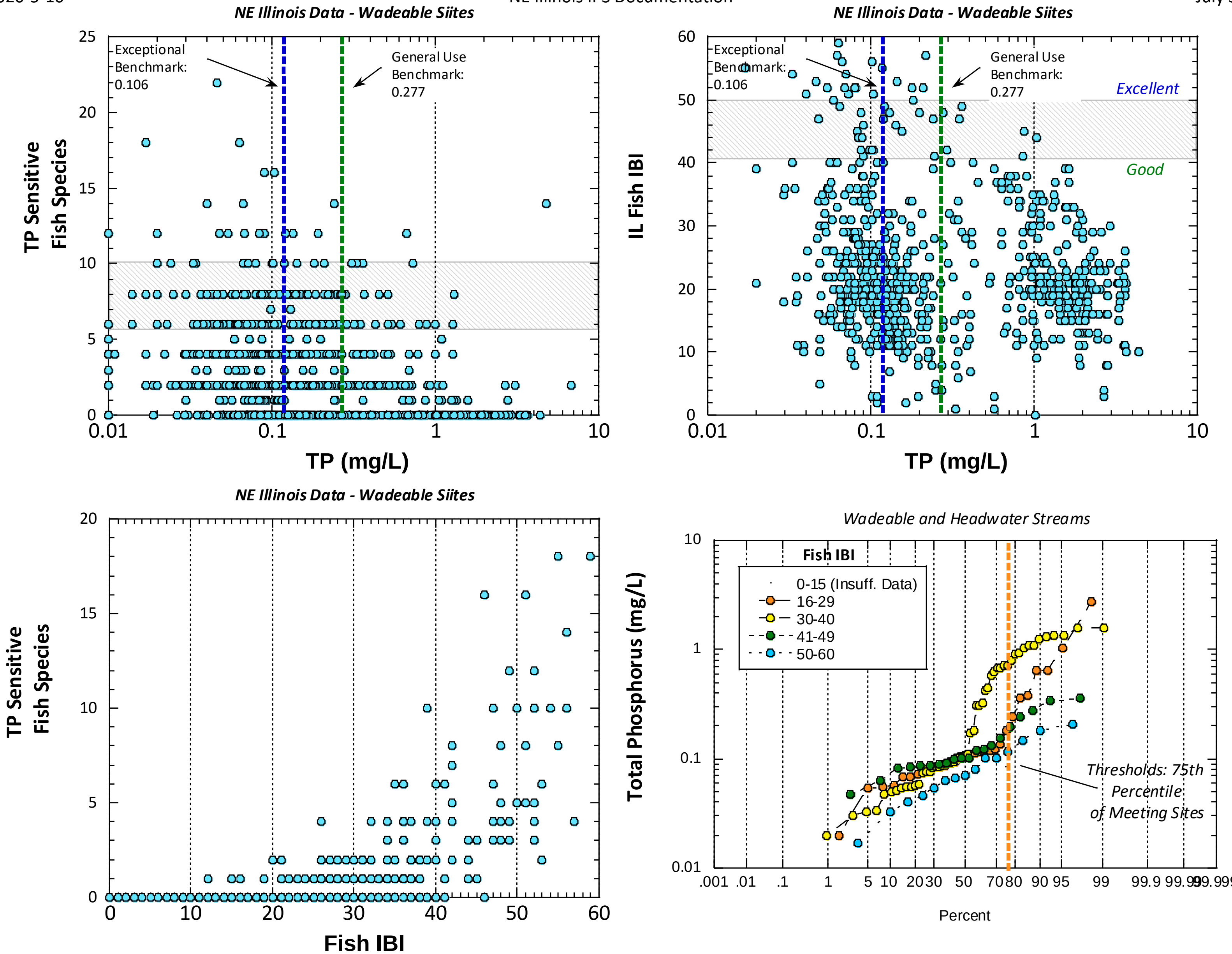

***Appendix Figure B-23**. Plots supporting derivation to $BOD_5$ benchmarks for wadeable streams in NE Illinois including scatter plots of $BOD_5$ vs $BOD_5$-macroinvertebrate taxa (top left), IBI vs. $BOD_5$-sensitive macroinvertebrate taxa (bottom left), $BOD_5$ vs. mIBI (top right) and a probability plot of $BOD_5$ values by narrative ranges of the mIBI. Data from IL IPS study area sites in NE Illinois (see text).*



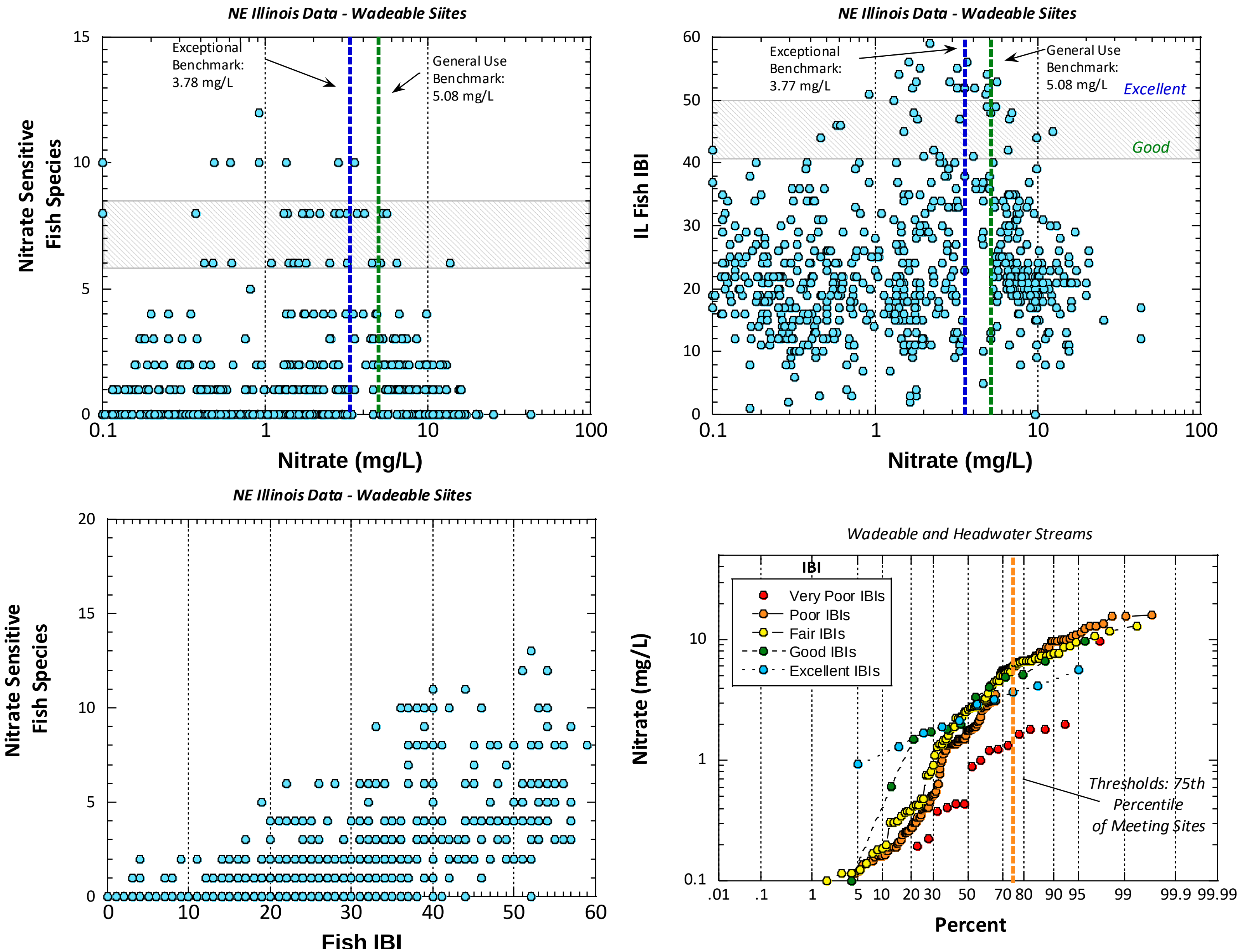

***Appendix Figure B-24****. Plots supporting derivation to total ammonia benchmarks for wadeable streams in NE Illinois including scatter plots of total ammonia vs total ammonia -macroinvertebrate taxa (top left), IBI vs. total ammonia-sensitive macroinvertebrate taxa (bottom left), total ammonia vs. mIBI (top right) and a probability plot of total ammonia values by narrative ranges of the mIBI. Data from IL IPS study area sites in NE Illinois (see text).*



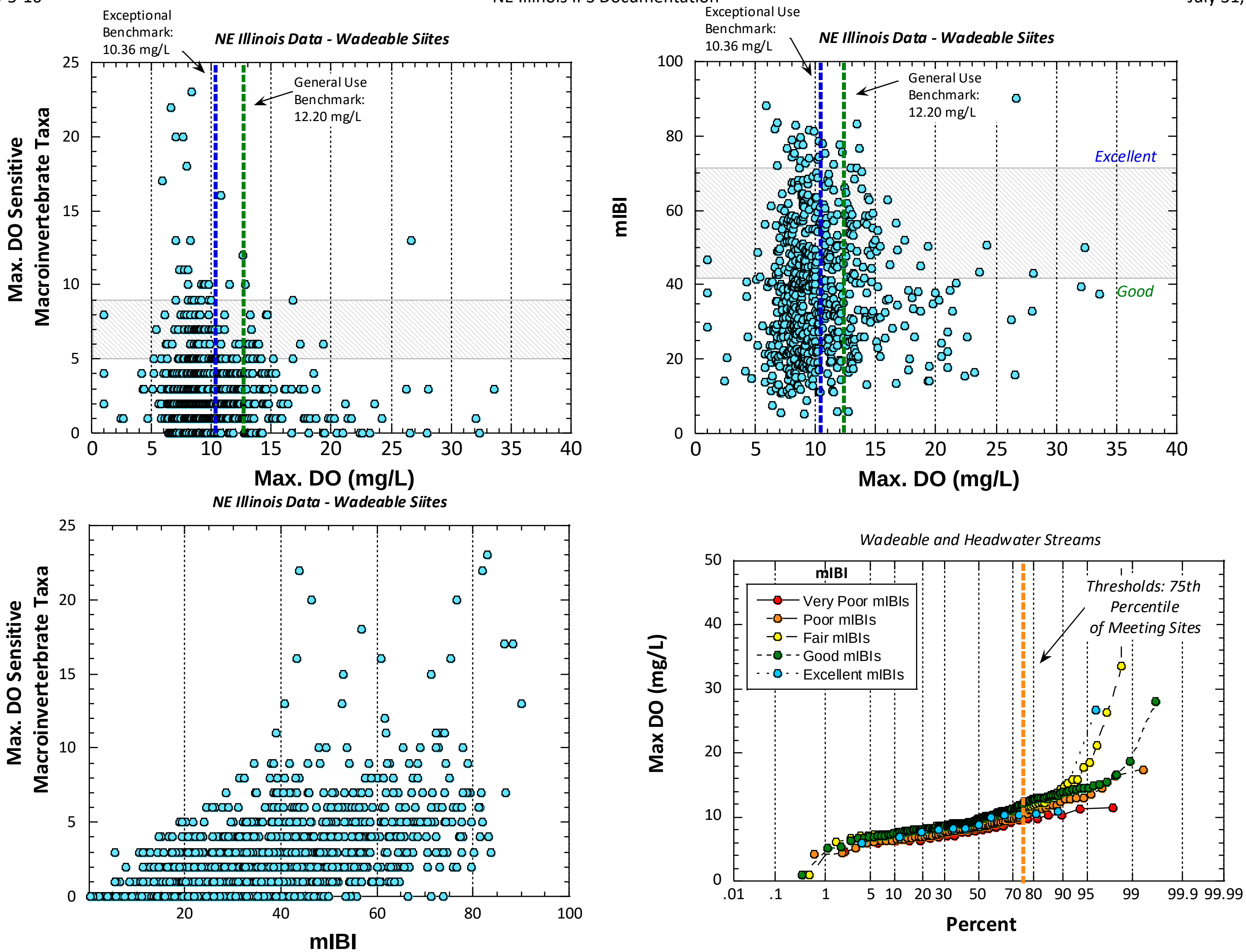

***Appendix Figure B-25**. Plots supporting derivation to TKN benchmarks for wadeable streams in NE Illinois including scatter plots of TKN vs TKN sensitive -macroinvertebrate taxa (top left), IBI vs. TKN-sensitive macroinvertebrate taxa (bottom left), TKN vs. mIBI (top right) and a probability plot of TKN values by narrative ranges of the mIBI. Data from IL IPS study area sites in NE Illinois (see text).*



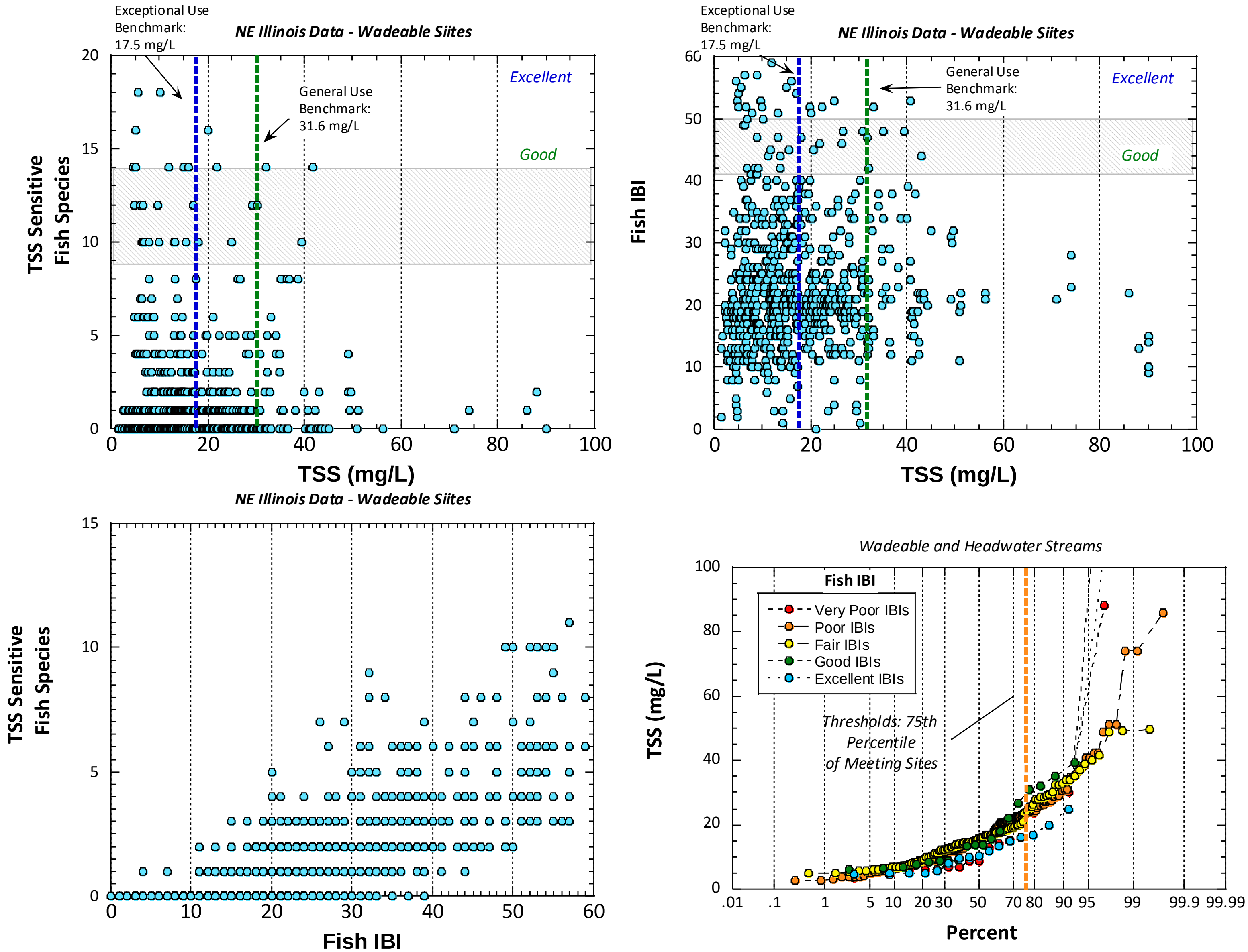

***Appendix Figure B-26**. Plots supporting derivation to TP benchmarks for wadeable streams in NE Illinois including scatter plots of TP vs TP sensitive fish species (top left), IBI vs. TP-sensitive fish species (bottom left), TP vs. IBI (top right) and a probability plot of TP values by narrative ranges of the fish IBI. Data from IL IPS study area sites in NE Illinois (see text).*



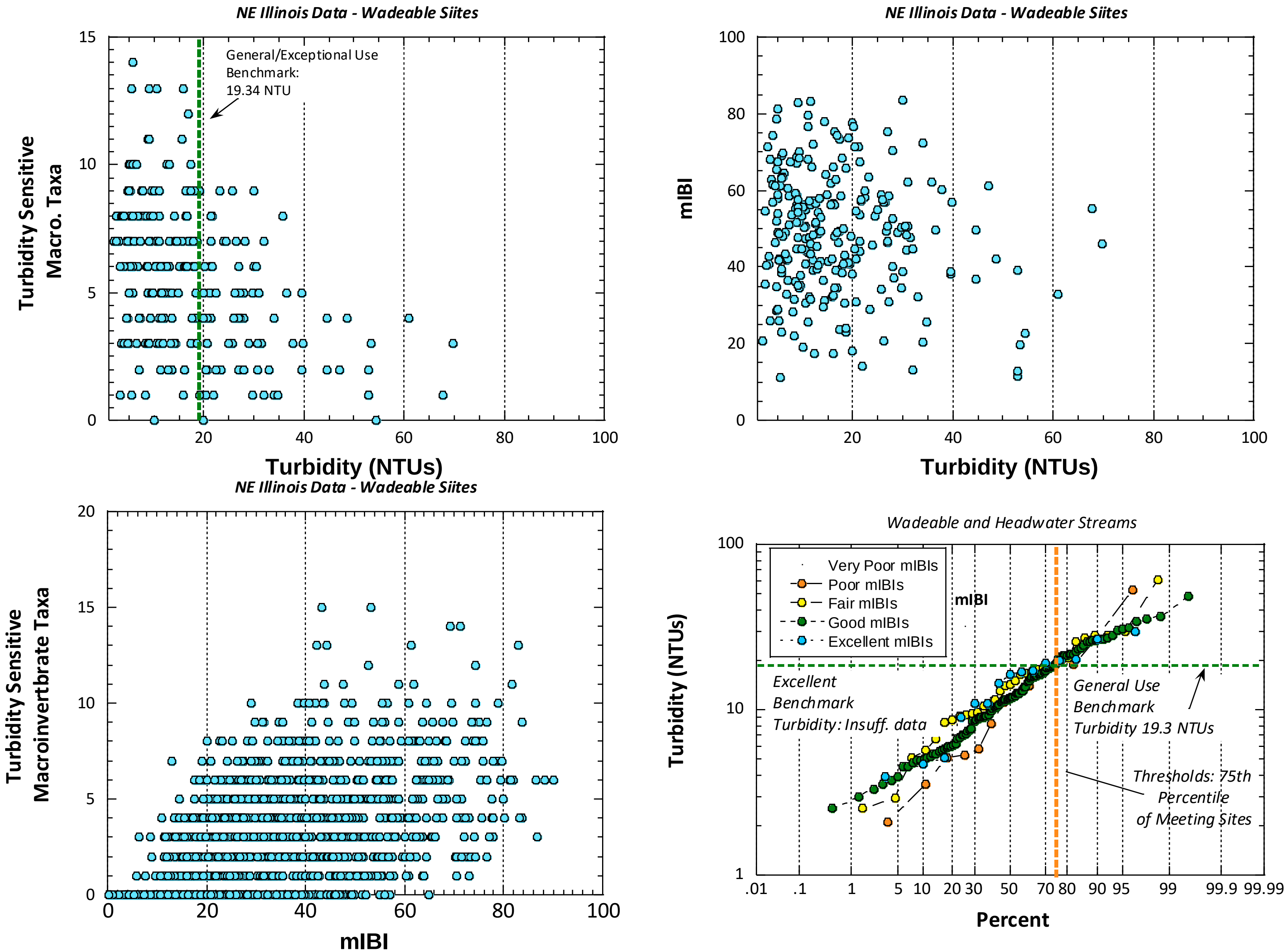

***Appendix Figure B-27****. Plots supporting derivation to nitrate benchmarks for wadeable streams in NE Illinois including scatter plots of nitrate vs nitrate sensitive fish species (top left), IBI vs. nitrate -sensitive fish species (bottom left), nitrate vs. IBI (top right) and a probability plot of nitrate values by narrative ranges of the fish IBI. Data from IL IPS study area sites in NE Illinois (see text).*



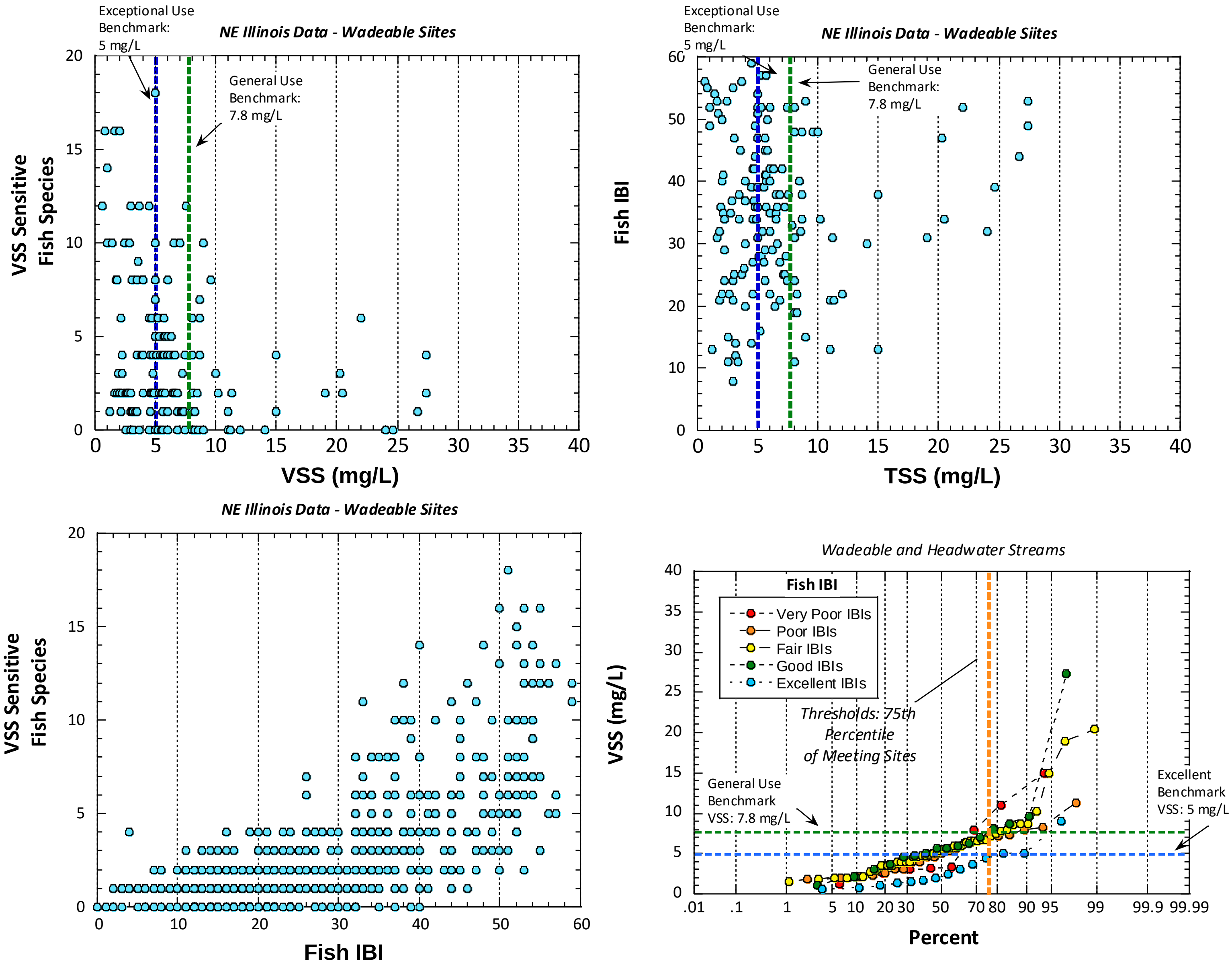

***Appendix Figure B-28.*** *Plots supporting derivation to max. DO benchmarks for wadeable streams in NE Illinois including scatter plots of max. DO vs max. DO sensitive macroinvertebrate taxa (top left), mIBI vs. max. DO -sensitive macroinvertebrate taxa (bottom left), max. DO vs. mIBI (top right) and a probability plot of max. DO values by narrative ranges of the mIBI. Data from IL IPS study area sites in NE Illinois (see text).*



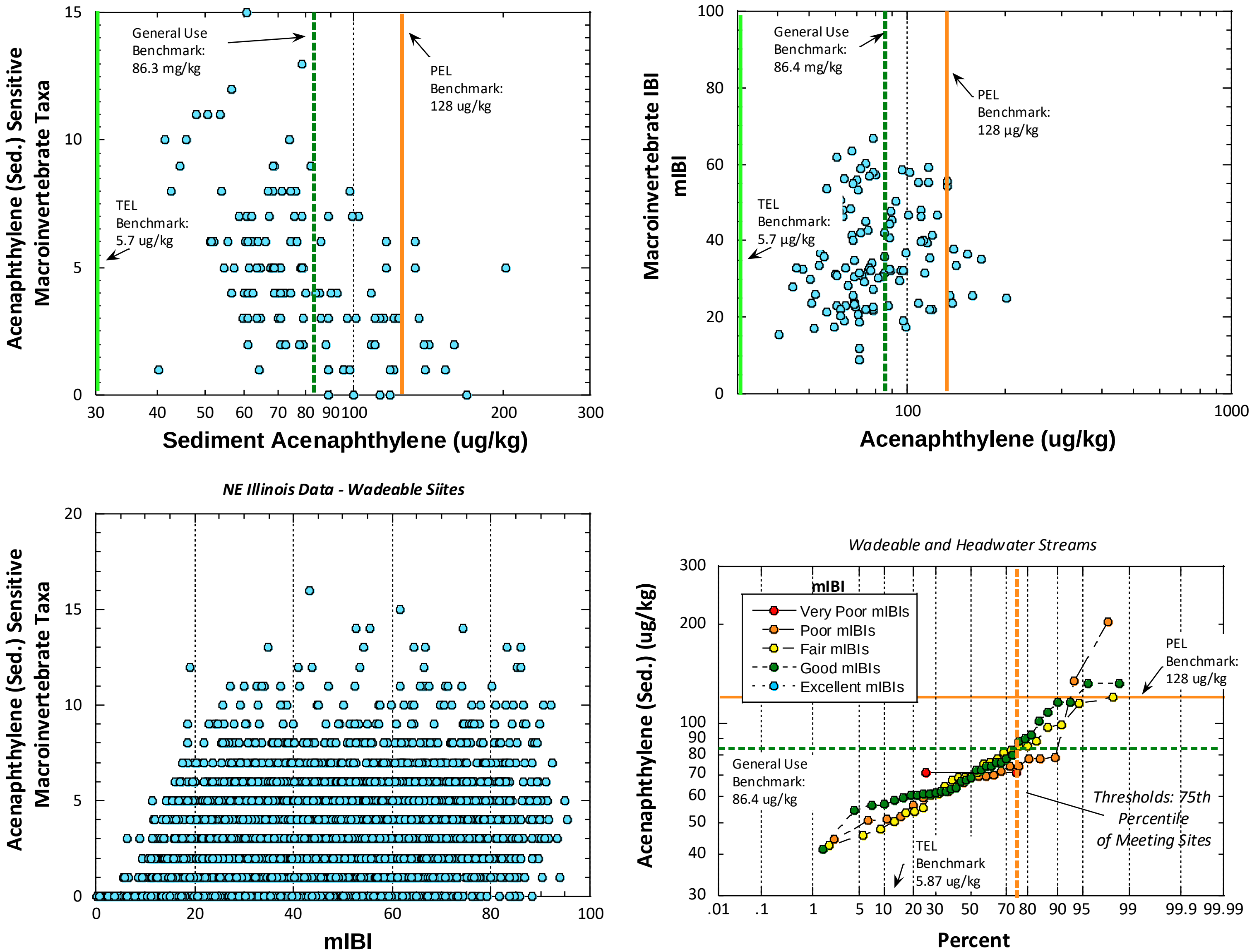

***Appendix Figure B-29*** *Plots supporting derivation to TSS benchmarks for wadeable streams in NE Illinois including scatter plots of TSS vs TSS sensitive fish species (top left), IBI vs. TSS -sensitive fish species (bottom left), TSS vs. IBI (top right) and a probability plot of TSS values by narrative ranges of the fish IBI. Data from IL IPS study area sites in NE Illinois (see text).*



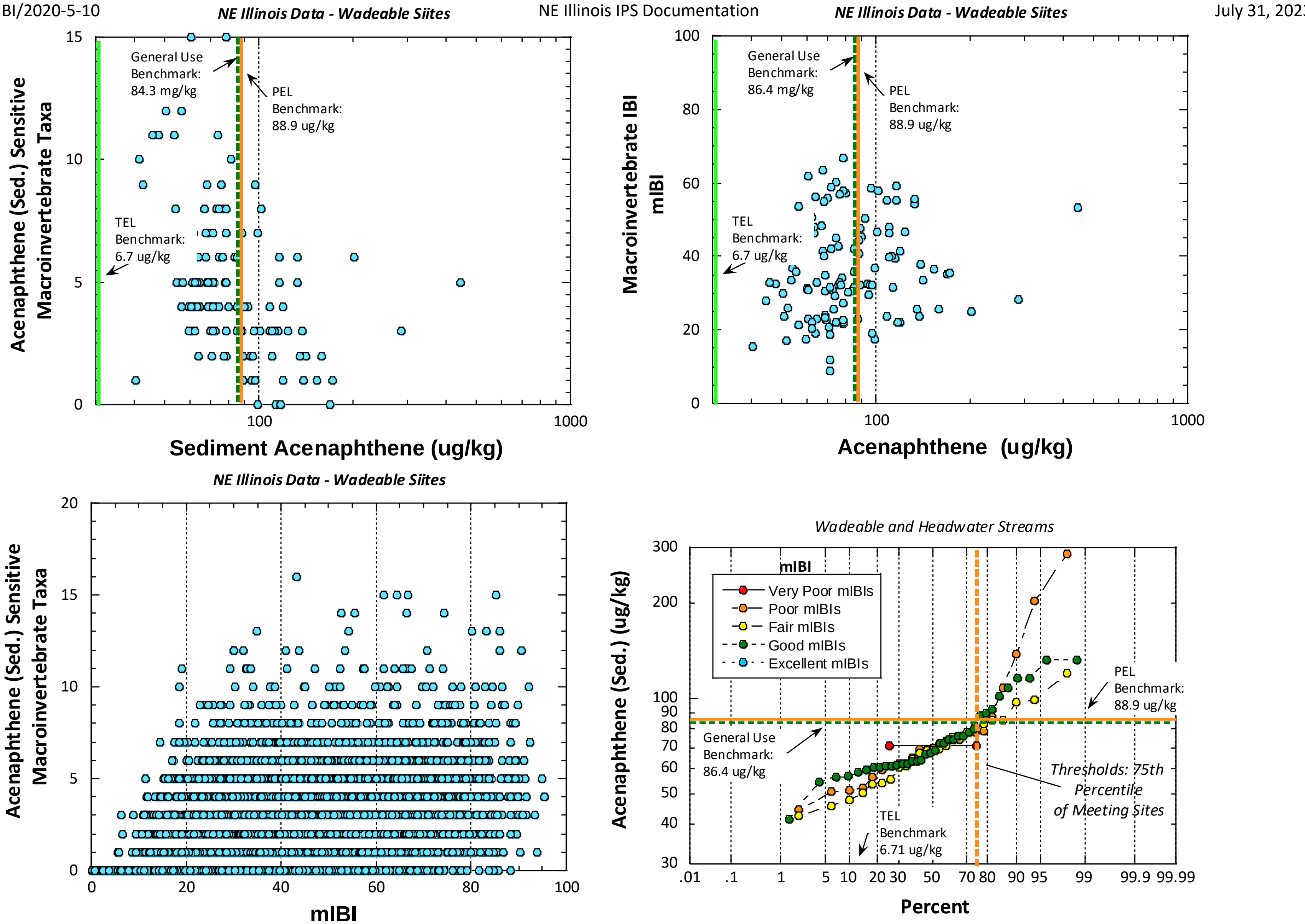

***Appendix Figure B-30.*** *Plots supporting derivation to turbidity benchmarks for wadeable streams in NE Illinois including scatter plots of turbidity vs turbidity sensitive macroinvertebrate taxa (top left), mIBI vs. turbidity -sensitive macroinvertebrate taxa (bottom left), turbidity vs. mIBI (top right) and a probability plot of turbidity values by narrative ranges of the mIBI. Data from IL IPS study area sites in NE Illinois (see text).*



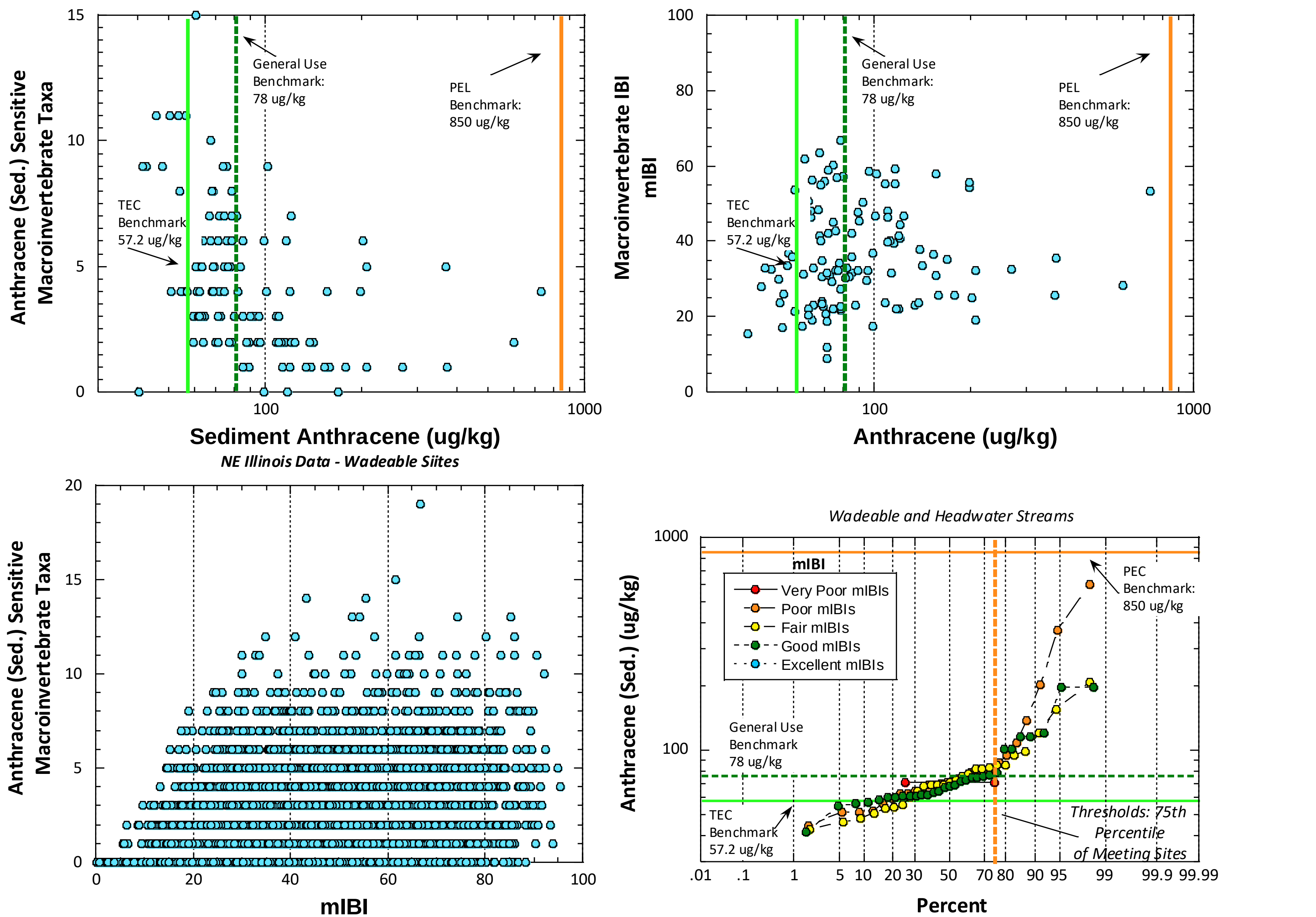

***Appendix Figure B-31.*** *Plots supporting derivation of VSS benchmarks for wadeable streams in NE Illinois including scatter plots of VSS vs VSS sensitive fish species (top left), IBI vs. VSS -sensitive fish species (bottom left), VSS vs. fish IBI (top right) and a probability plot of VSS values by narrative ranges of the IBI. Data from IL IPS study area sites in NE Illinois (see text).*



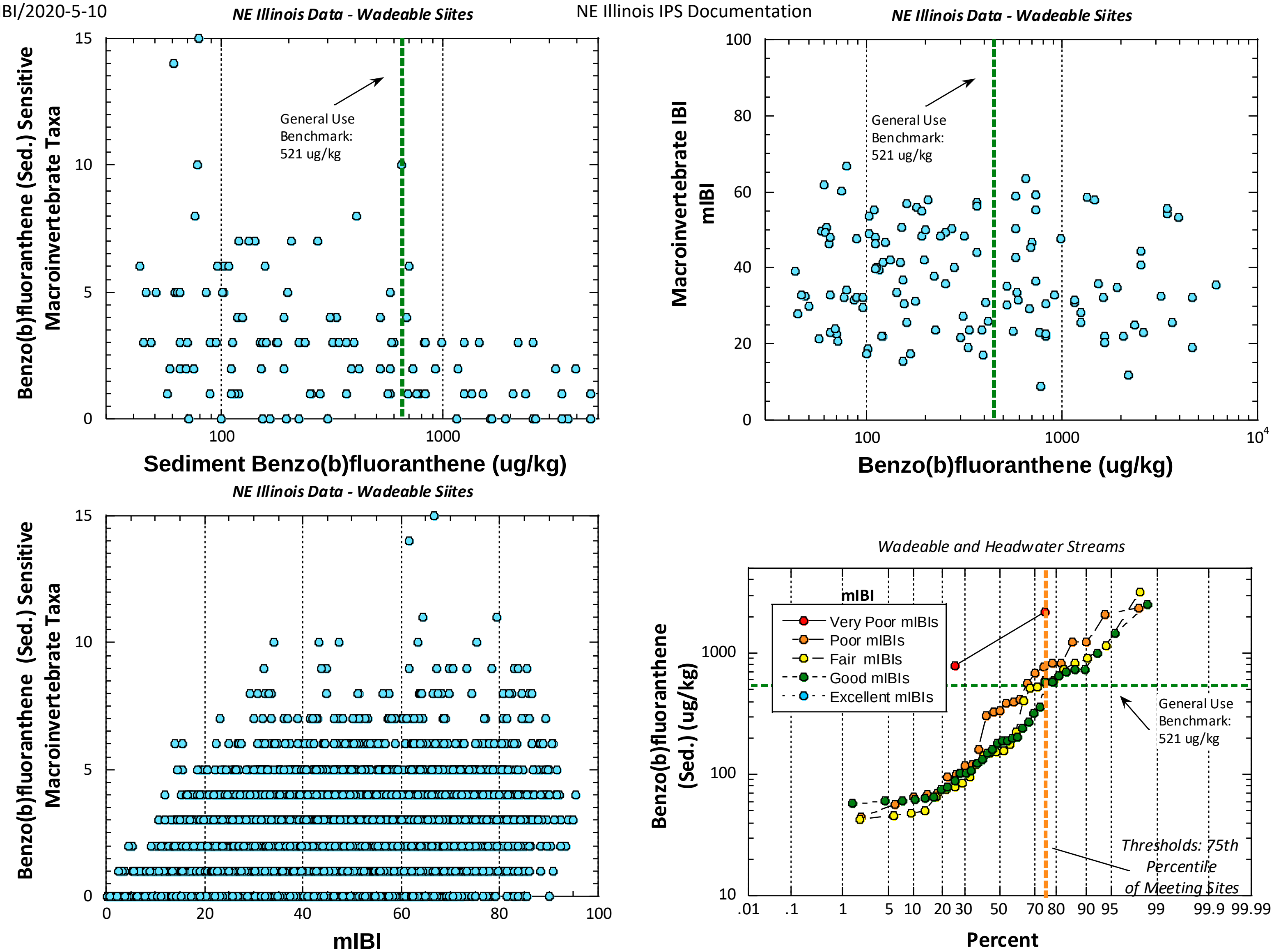

***Appendix Figure B-32.*** *Plots supporting derivation to acenapthylene (sediment) benchmarks for wadeable streams in NE Illinois including scatter plots of acenapthylene vs acenapthylene sensitive macroinvertebrate taxa (top left), mIBI vs. acenapthylene -sensitive macroinvertebrate taxa (bottom left), acenapthylene vs. mIBI (top right) and a probability plot of acenapthylene values by narrative ranges of the mIBI. Various benchmarks are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*



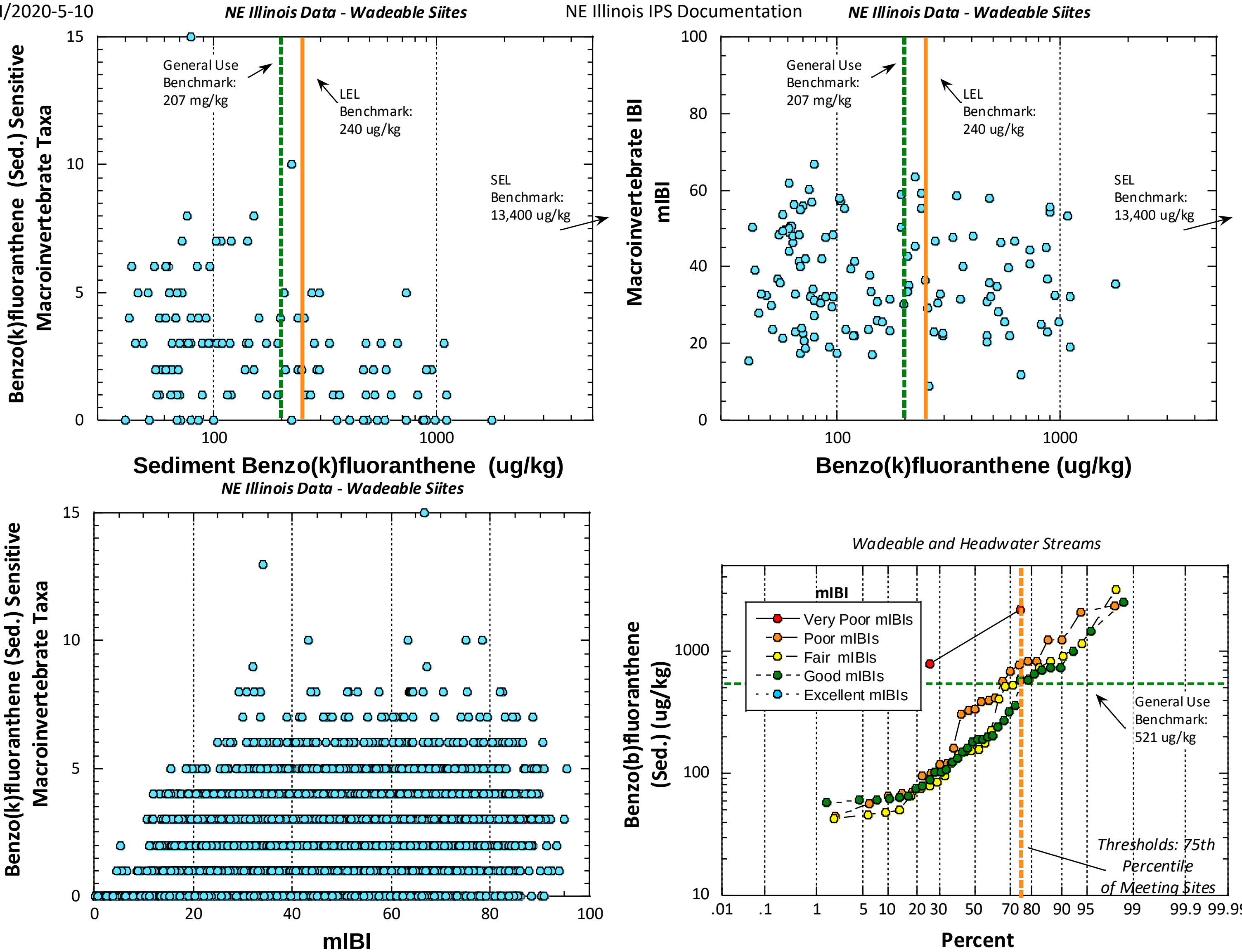

***Appendix Figure B-33.*** *Plots supporting derivation to acenaphthene (sediment) benchmarks for wadeable streams in NE Illinois including scatter plots of acenaphthene vs acenaphthene sensitive macroinvertebrate taxa (top left), mIBI vs. acenaphthene-sensitive macroinvertebrate taxa (bottom left), acenaphthene vs. mIBI (top right) and a probability plot of acenaphthene values by narrative ranges of the mIBI. Various benchmarks are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*



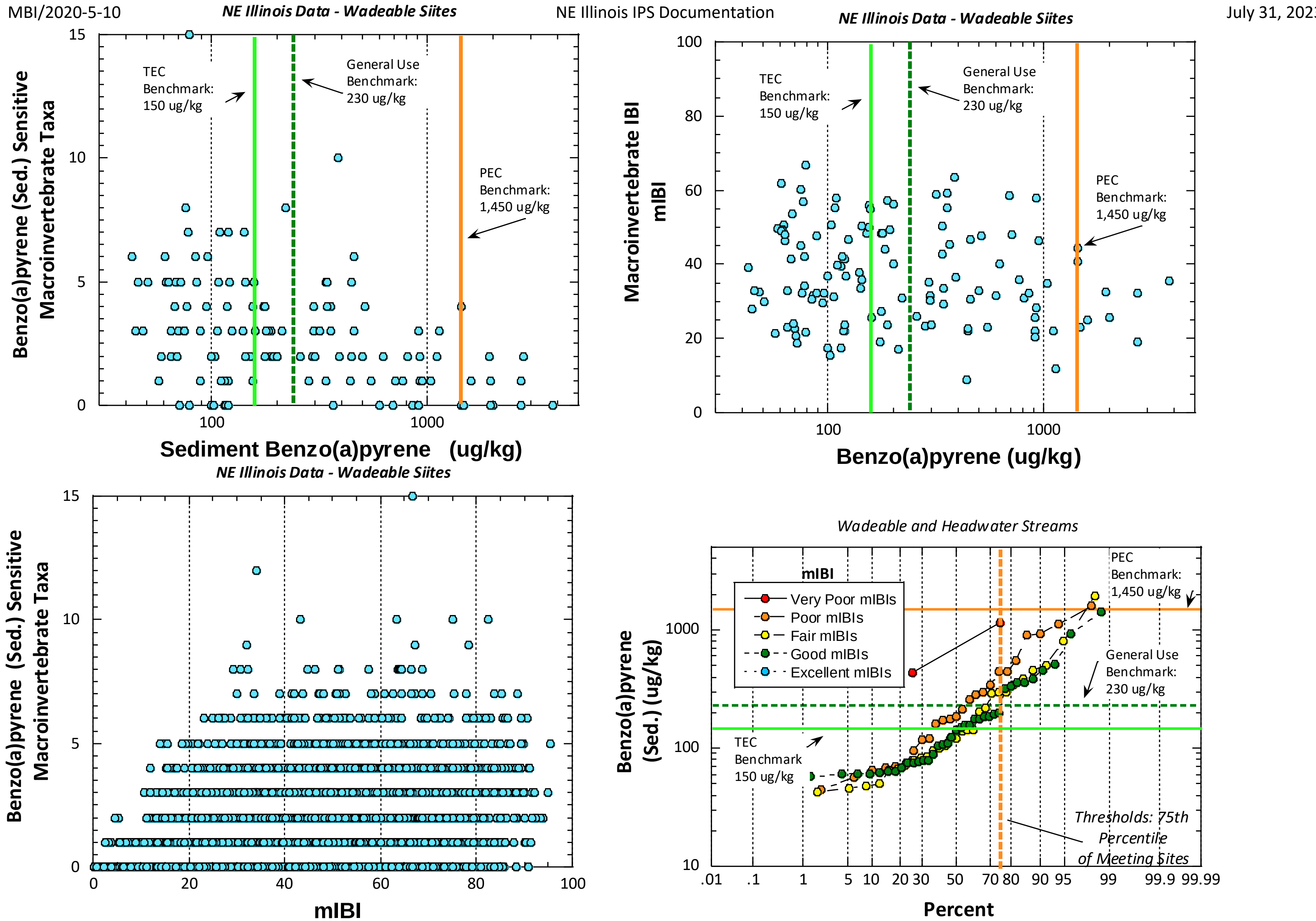

***Appendix Figure B-34.*** *Plots supporting derivation to anthracene (sediment) benchmarks for wadeable streams in NE Illinois including scatter plots of anthracene vs anthracene sensitive macroinvertebrate taxa (top left), mIBI vs. anthracene -sensitive macroinvertebrate taxa (bottom left), anthracene vs. mIBI (top right) and a probability plot of anthracene values by narrative ranges of the mIBI. Various benchmarks are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*



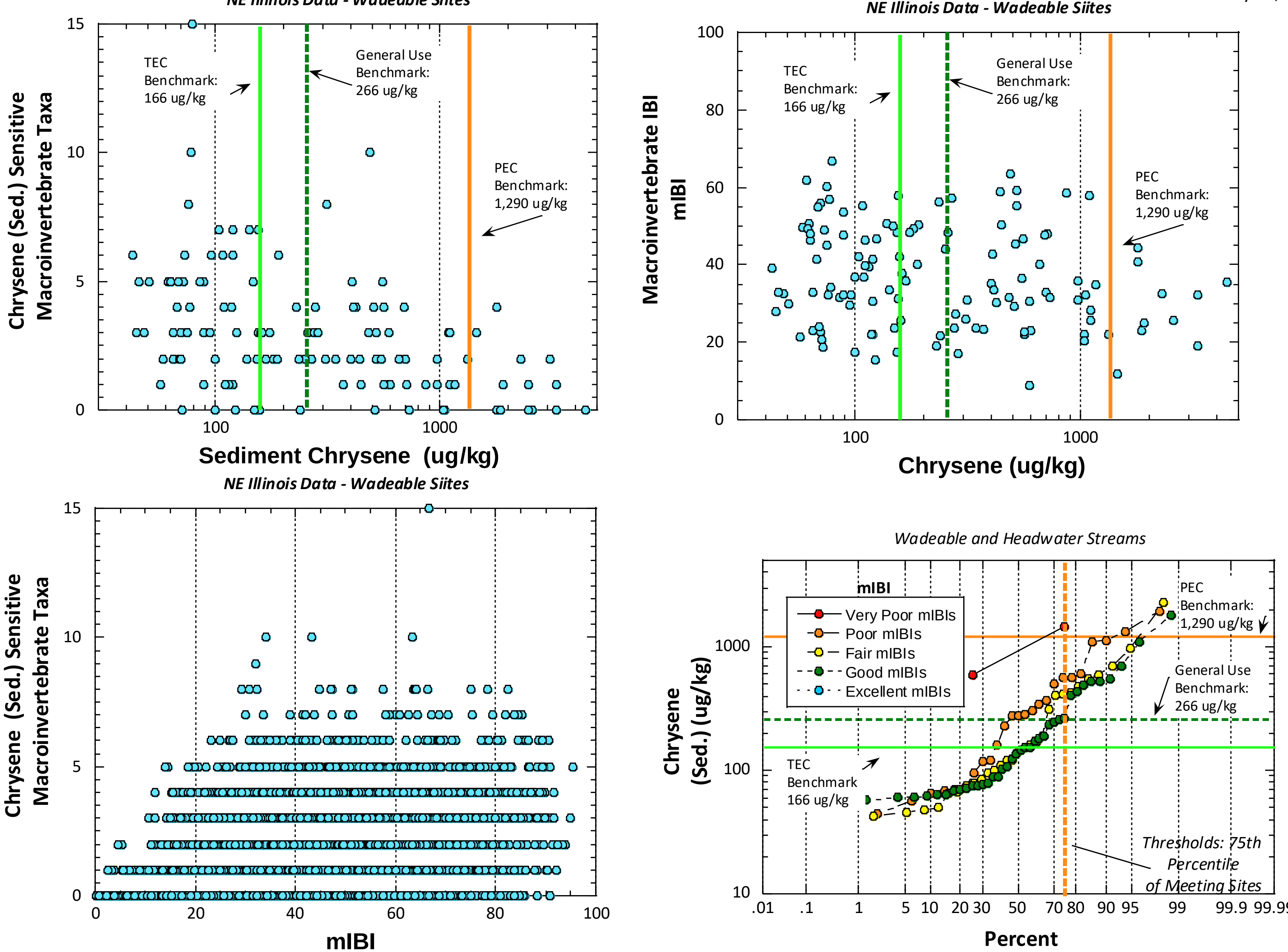

***Appendix Figure B-35.*** *Plots supporting derivation to benzo(b)fluoranthene (sediment) benchmarks for wadeable streams in NE Illinois including scatter plots of benzo(b)fluoranthene vs benzo(b)fluoranthene sensitive macroinvertebrate taxa (top left), mIBI vs. benzo(b)fluoranthene -sensitive macroinvertebrate taxa (bottom left), benzo(b)fluoranthene vs. mIBI (top right) and a probability plot of benzo(b)fluoranthene values by narrative ranges of the mIBI. Various benchmarks are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*



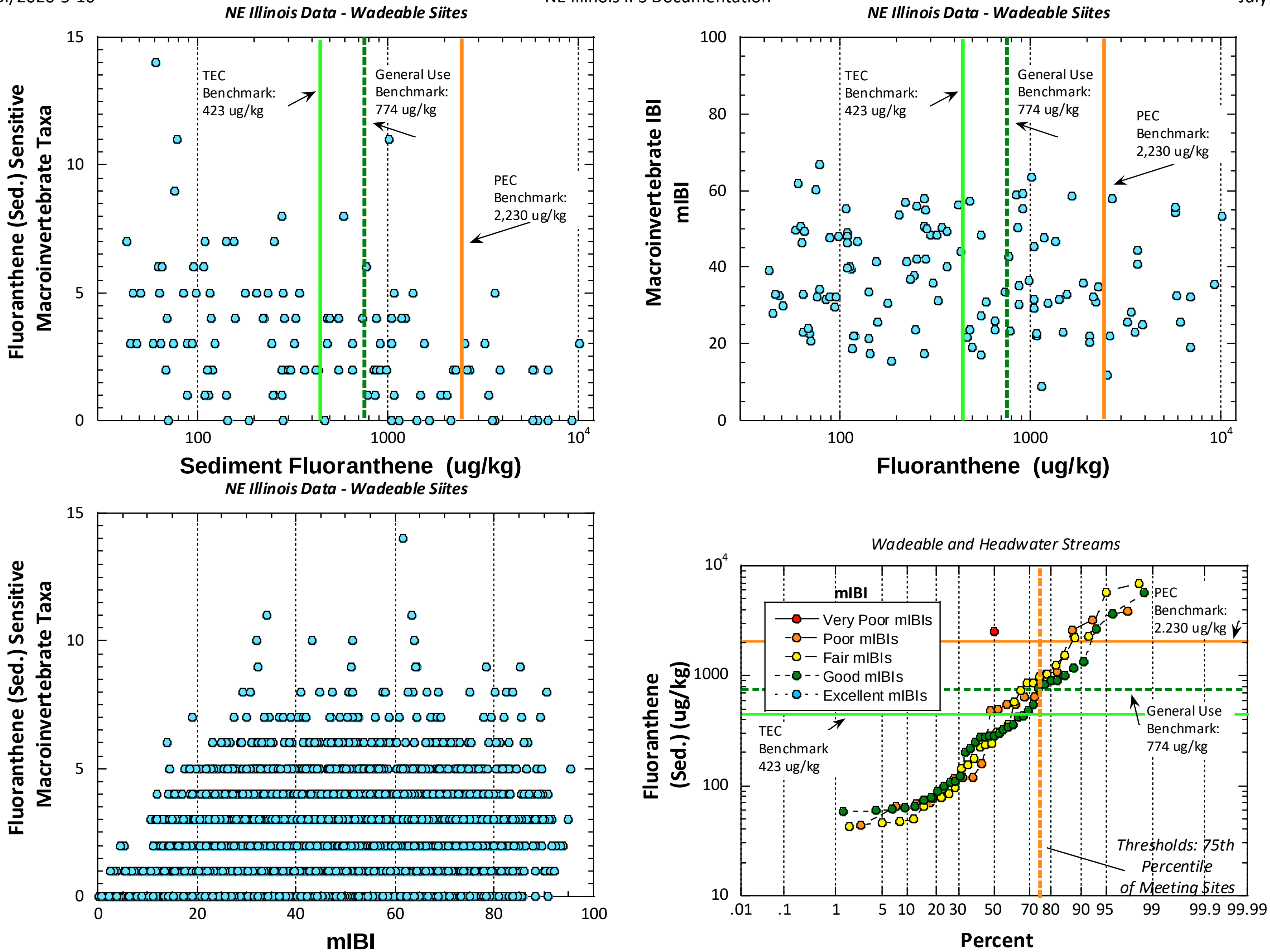

***Appendix Figure B-36.*** *Plots supporting derivation to Benzo(k)fluoranthene (sediment) benchmarks for wadeable streams in NE Illinois including scatter plots of benzo(k)fluoranthene vs benzo(k)fluoranthene sensitive macroinvertebrate taxa (top left), mIBI vs. benzo(k)fluoranthene -sensitive macroinvertebrate taxa (bottom left), benzo(k)fluoranthene vs. mIBI (top right) and a probability plot of benzo(k)fluoranthene values by narrative ranges of the mIBI. Various benchmarks are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*



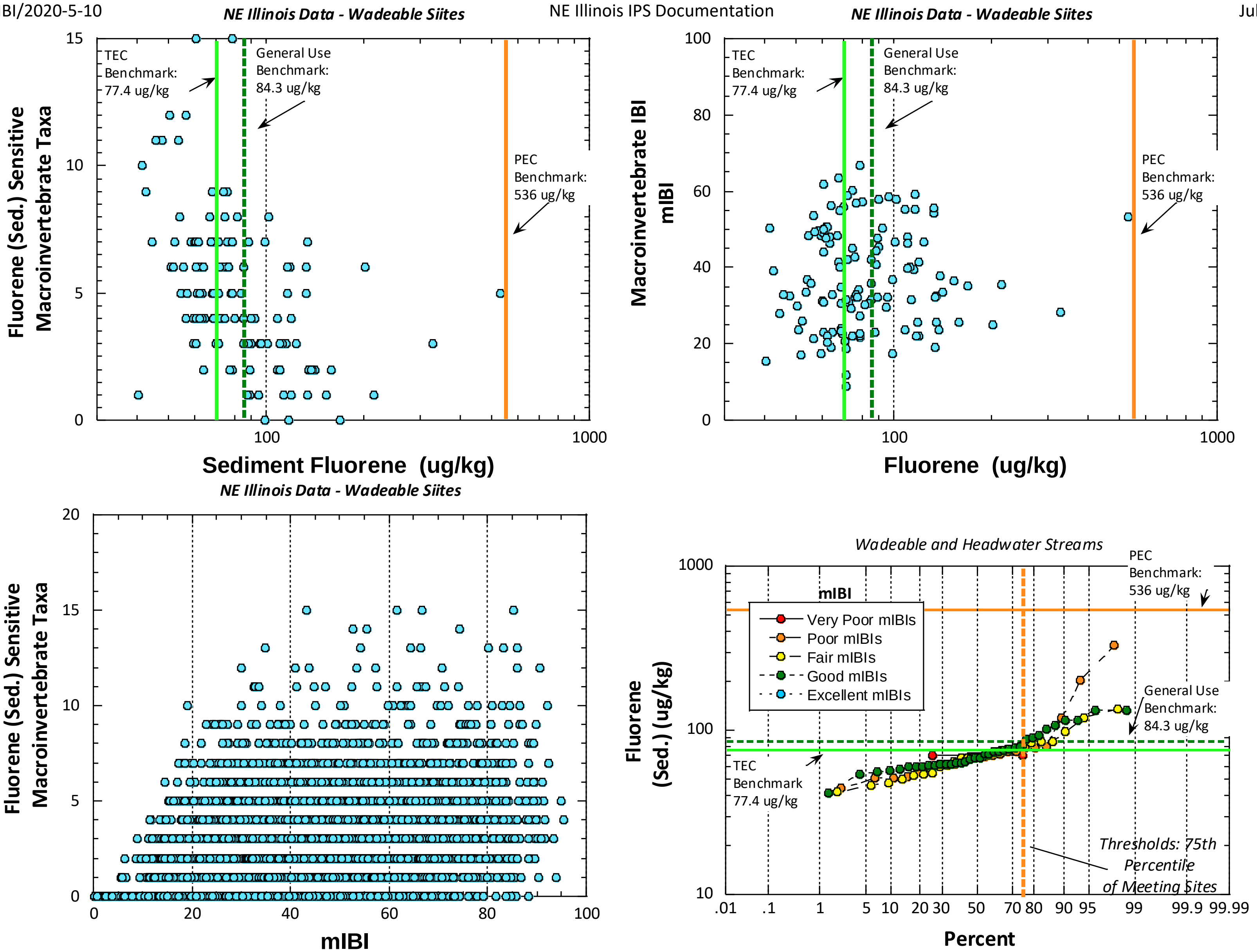

***Appendix Figure B-37.*** *Plots supporting derivation to benzo(a)pyrene (sediment) benchmarks for wadeable streams in NE Illinois including scatter plots of benzo(a)pyrene vs benzo(a)pyrene sensitive macroinvertebrate taxa (top left), mIBI vs. benzo(a)pyrene -sensitive macroinvertebrate taxa (bottom left), benzo(a)pyrene vs. mIBI (top right) and a probability plot of benzo(a)pyrene values by narrative ranges of the mIBI. Various benchmarks are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*



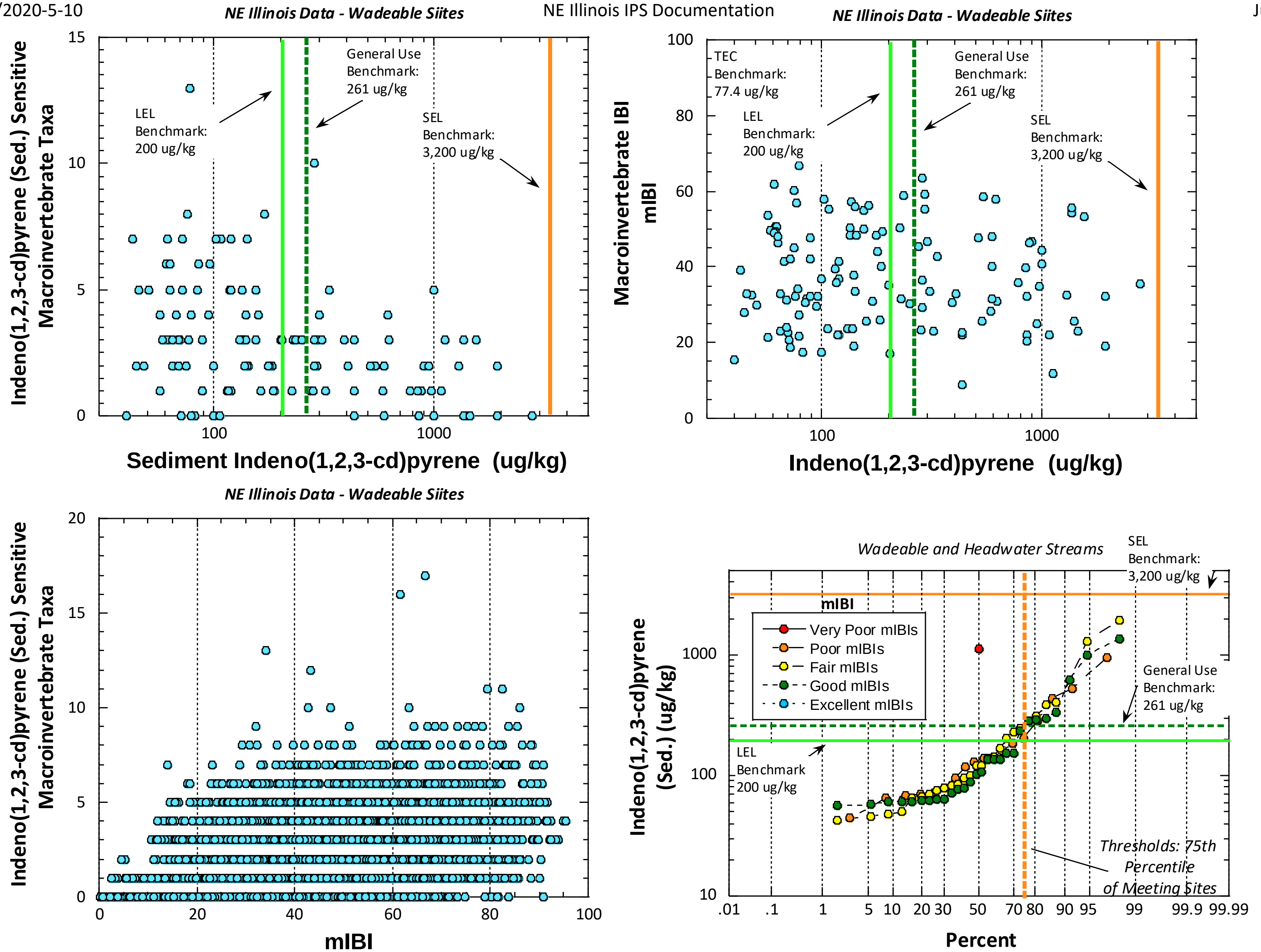

***Appendix Figure B-38.*** *Plots supporting derivation to chrysene (sediment) benchmarks for wadeable streams in NE Illinois including scatter plots of chrysene vs. chrysene sensitive macroinvertebrate taxa (top left), mIBI vs. chrysene -sensitive macroinvertebrate taxa (bottom left), chrysene vs. mIBI (top right) and a probability plot of chrysene values by narrative ranges of the mIBI. Various benchmarks are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*



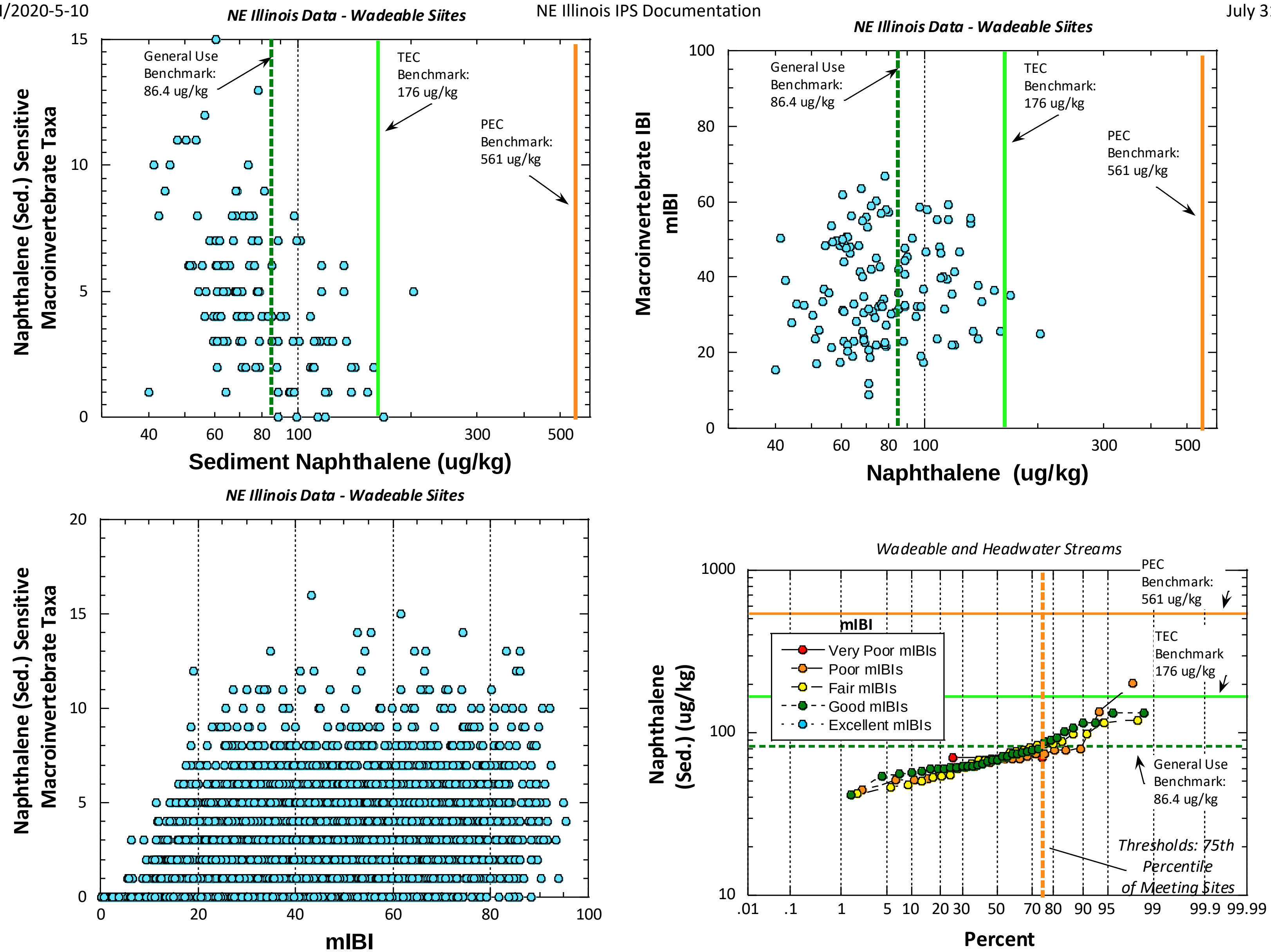

***Appendix Figure B-39.*** *Plots supporting derivation to fluoranthene (sediment) benchmarks for wadeable streams in NE Illinois including scatter plots of fluoranthene vs. fluoranthene sensitive macroinvertebrate taxa (top left), mIBI vs. fluoranthene -sensitive macroinvertebrate taxa (bottom left), fluoranthene vs. mIBI (top right) and a probability plot of fluoranthene values by narrative ranges of the mIBI. Various benchmarks are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*



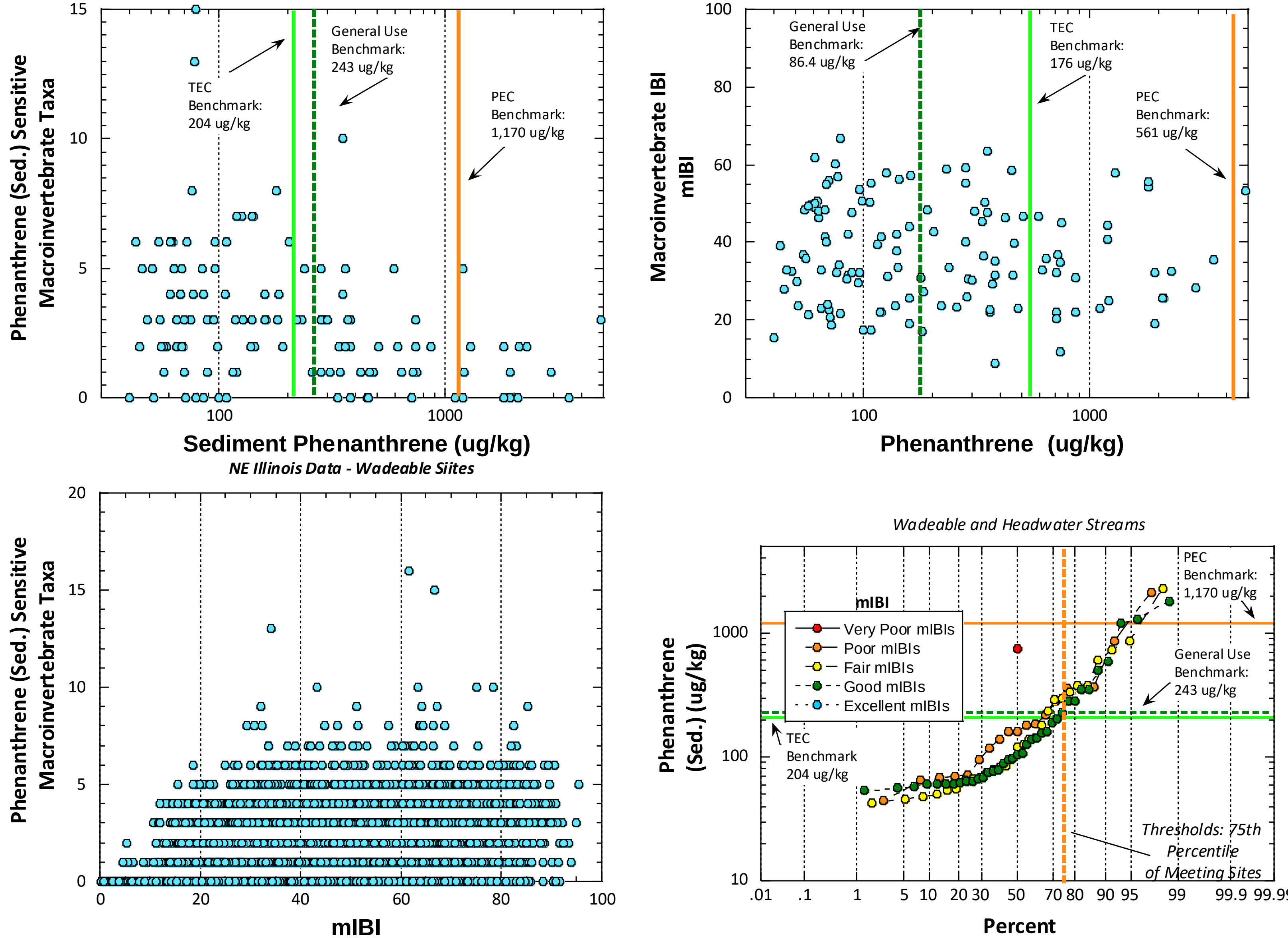

***Appendix Figure B-40.*** *Plots supporting derivation to fluorene (sediment) benchmarks for wadeable streams in NE Illinois including scatter plots of fluorene vs. fluorene sensitive macroinvertebrate taxa (top left), mIBI vs. fluorene -sensitive macroinvertebrate taxa (bottom left), fluorene vs. mIBI (top right) and a probability plot of fluorene values by narrative ranges of the mIBI. Various benchmarks are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*



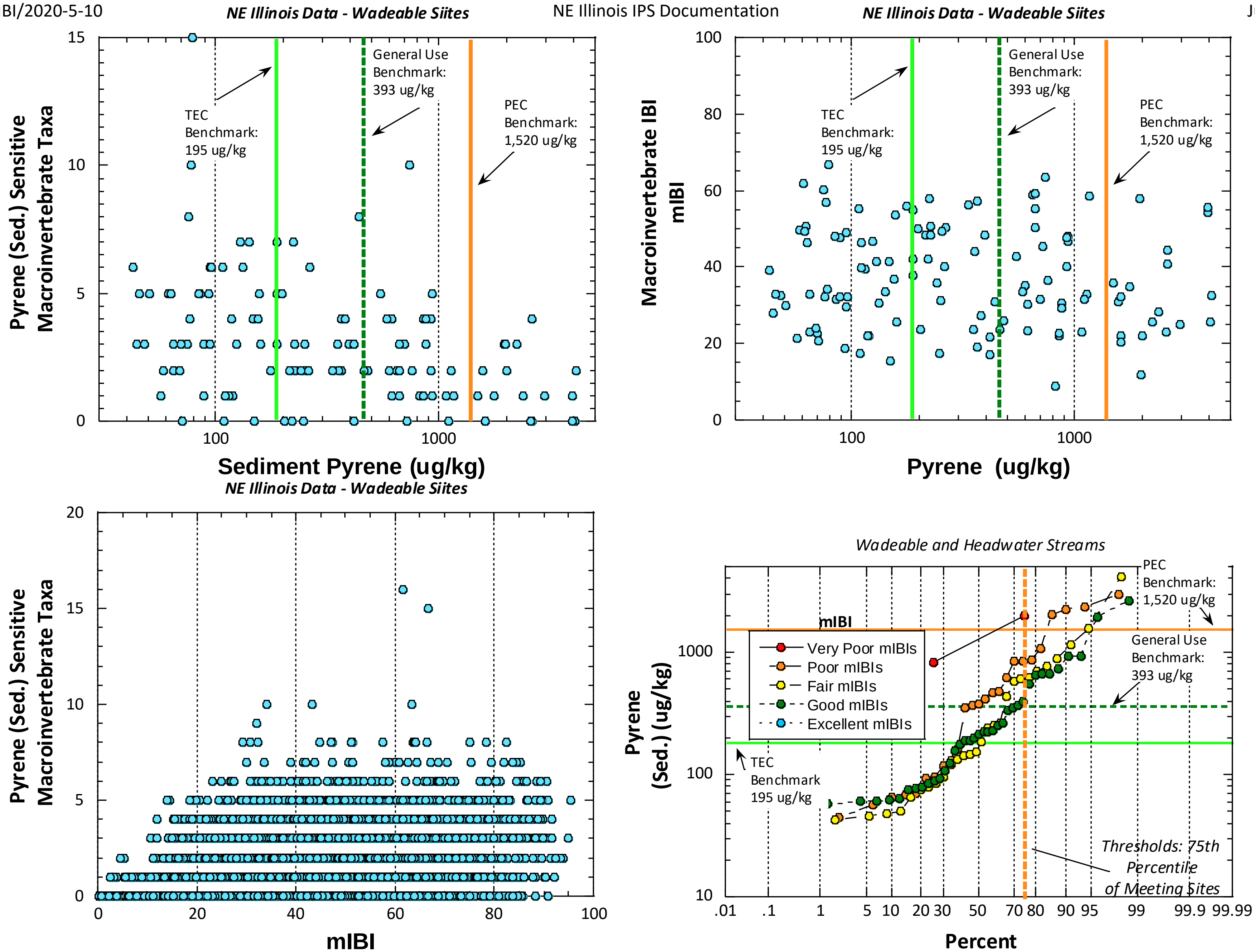

***Appendix Figure B-41.*** *Plots supporting derivation to indeno(1,2,3-cd)pyrene (sediment) benchmarks for wadeable streams in NE Illinois including scatter plots of indeno(1,2,3-cd)pyrene vs. indeno(1,2,3-cd)pyrene sensitive macroinvertebrate taxa (top left), mIBI vs. indeno(1,2,3-cd)pyrene -sensitive macroinvertebrate taxa (bottom left), indeno(1,2,3-cd)pyrene vs. mIBI (top right) and a probability plot of indeno(1,2,3-cd)pyrene values by narrative ranges of the mIBI. Various benchmarks are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*



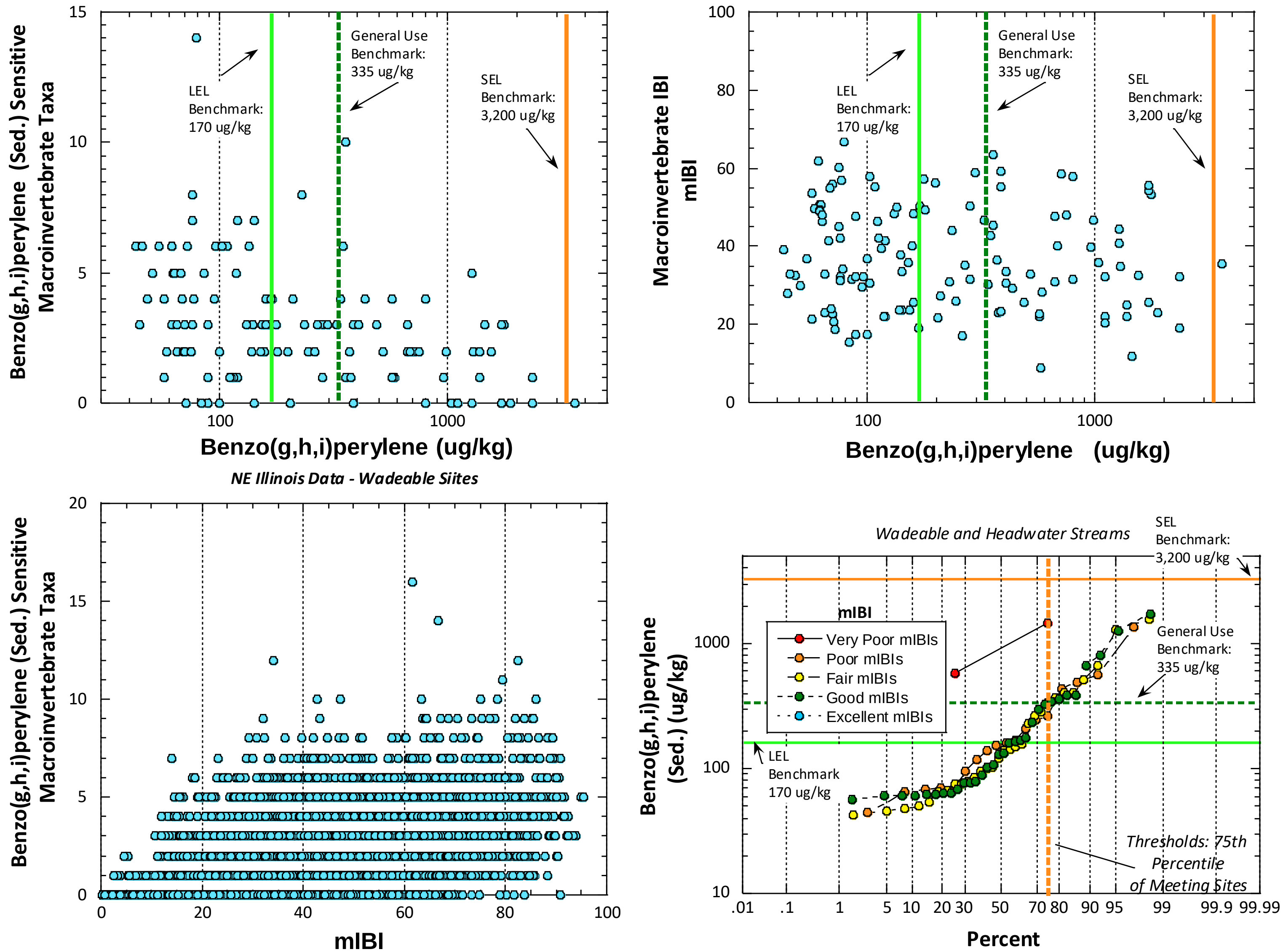

***Appendix Figure B-42.*** *Plots supporting derivation to naphthalene (sediment) benchmarks for wadeable streams in NE Illinois including scatter plots of naphthalene vs. naphthalene sensitive macroinvertebrate taxa (top left), mIBI vs. naphthalene -sensitive macroinvertebrate taxa (bottom left), naphthalene vs. mIBI (top right) and a probability plot of naphthalene values by narrative ranges of the mIBI. Various benchmarks are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*



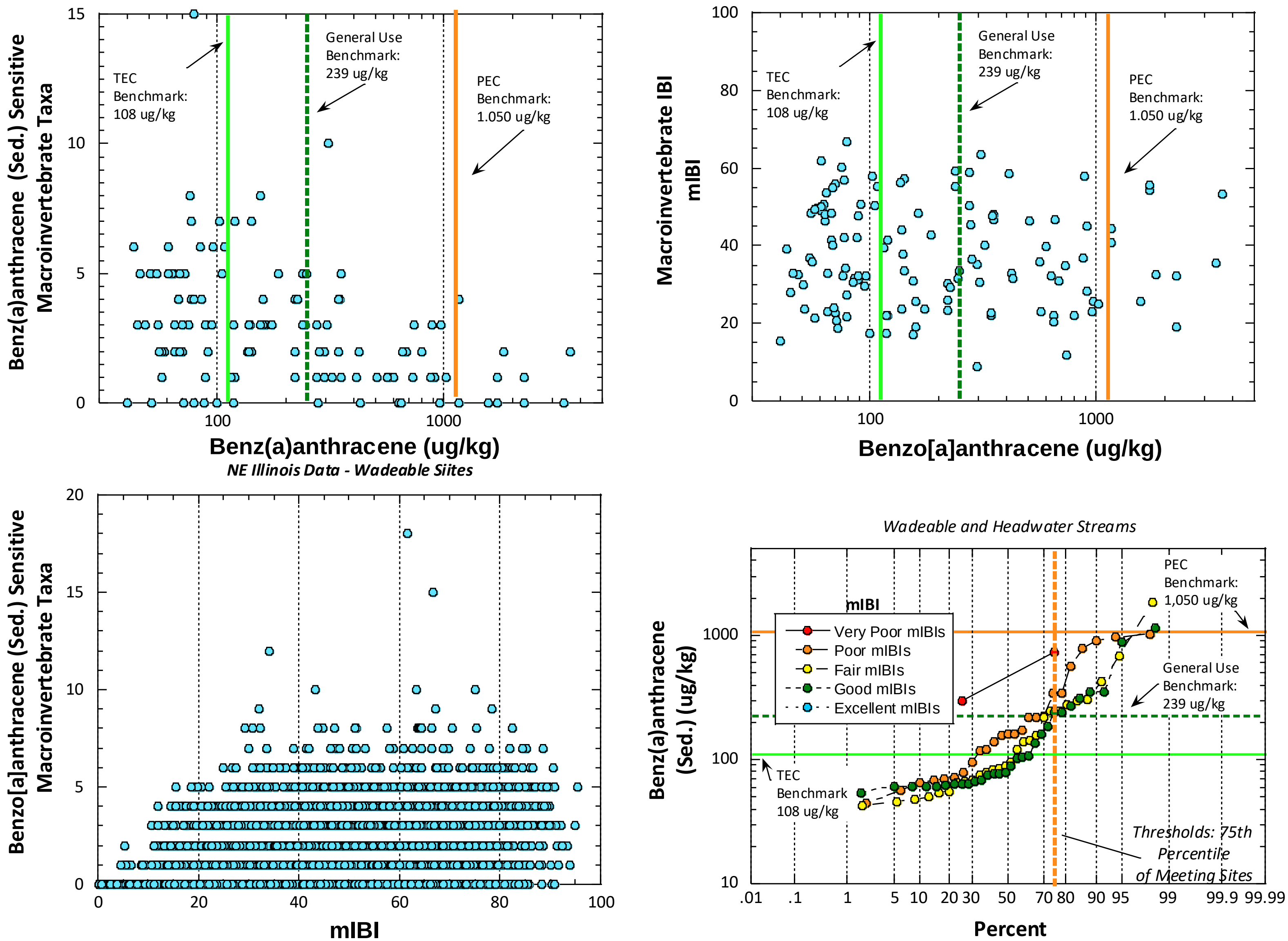

***Appendix Figure B-43.*** *Plots supporting derivation to phenanthrene (sediment) benchmarks for wadeable streams in NE Illinois including scatter plots of phenanthrene vs. phenanthrene sensitive macroinvertebrate taxa (top left), mIBI vs. phenanthrene -sensitive macroinvertebrate taxa (bottom left), phenanthrene vs. mIBI (top right) and a probability plot of phenanthrene values by narrative ranges of the mIBI. Various benchmarks are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*



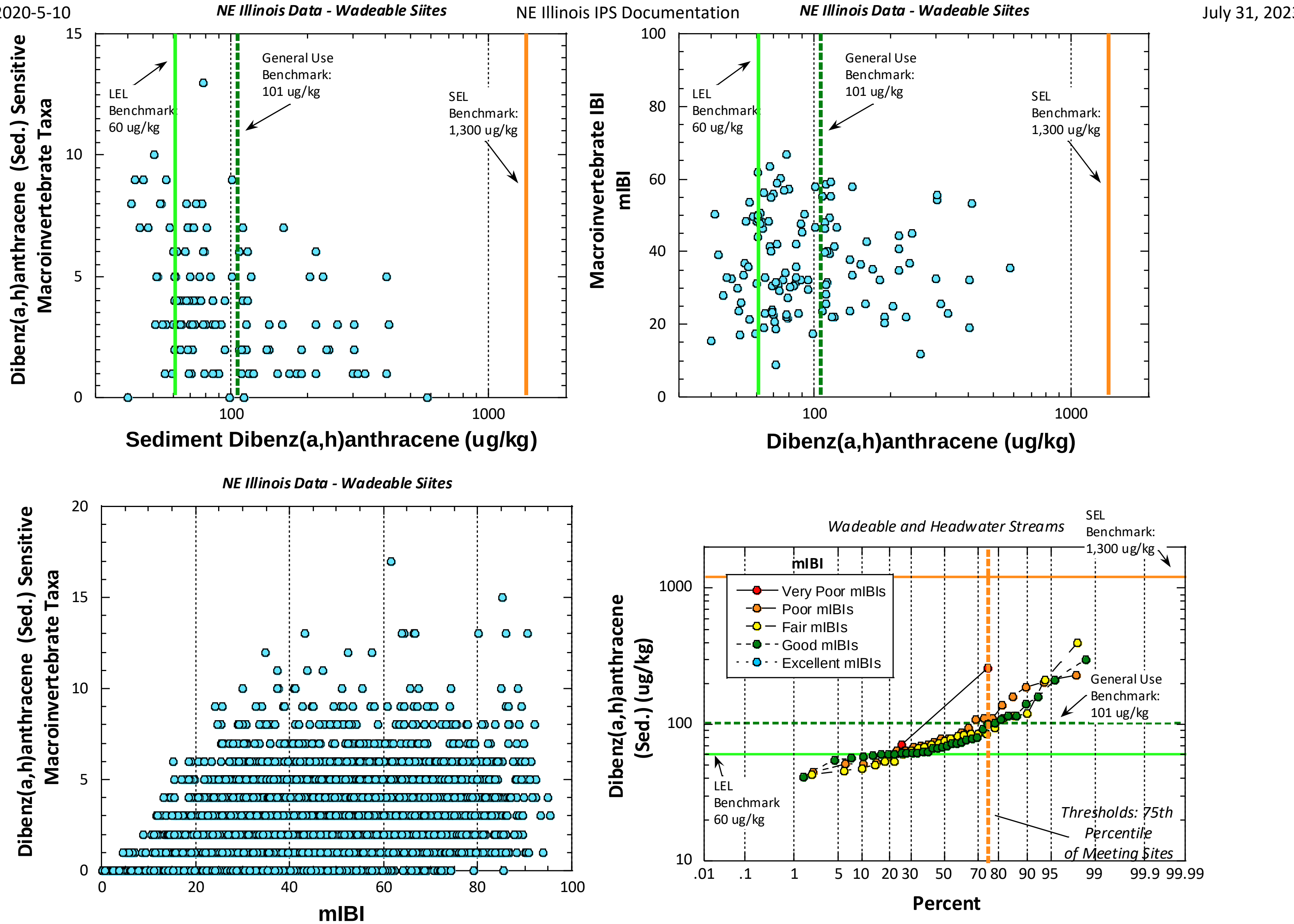

***Appendix Figure B-44.*** *Plots supporting derivation to pyrene (sediment) benchmarks for wadeable streams in NE Illinois including scatter plots of pyrene vs. pyrene sensitive macroinvertebrate taxa (top left), mIBI vs. pyrene -sensitive macroinvertebrate taxa (bottom left), pyrene vs. mIBI (top right) and a probability plot of pyrene values by narrative ranges of the mIBI. Various benchmarks are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*



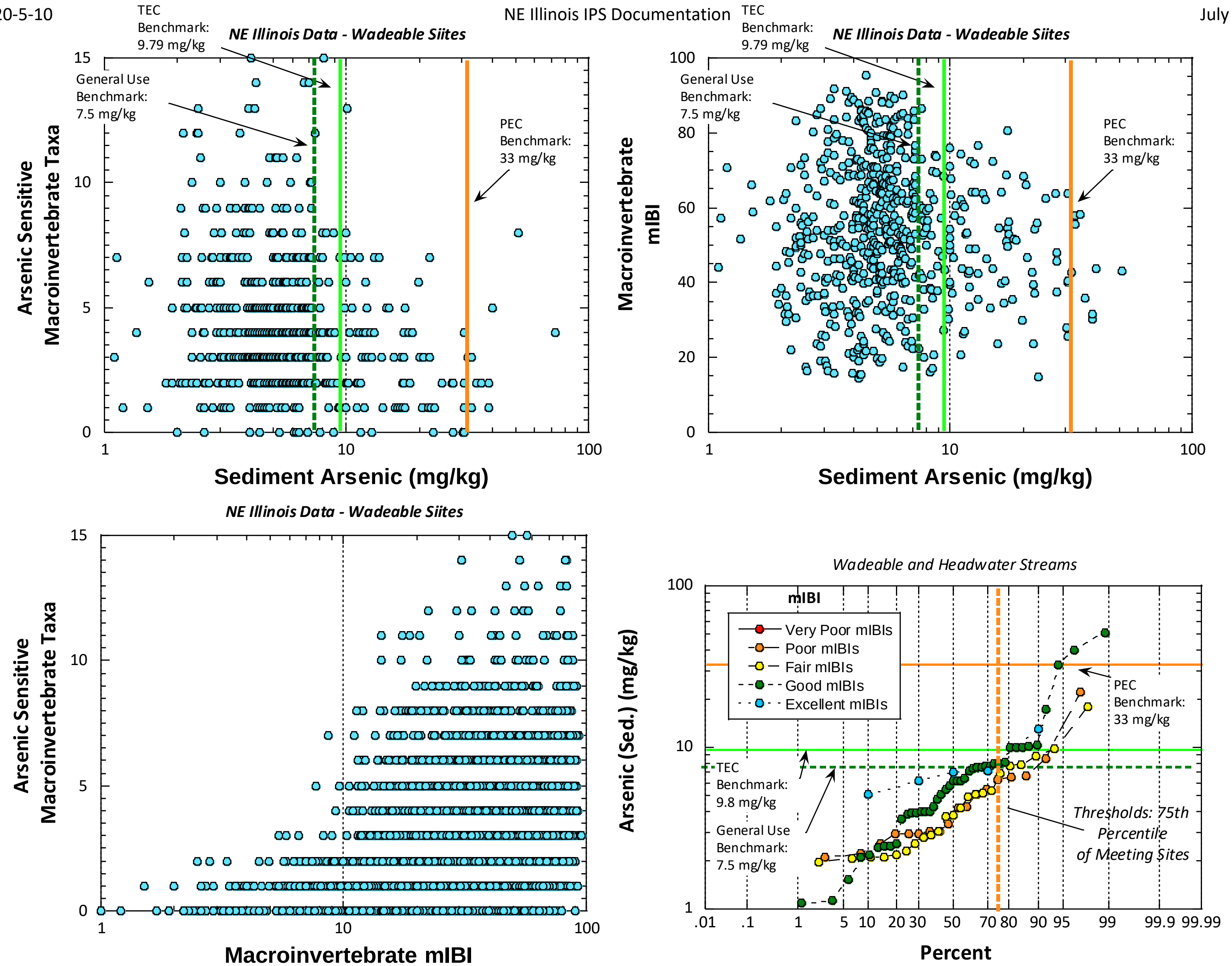

***Appendix Figure B-45.*** *Plots supporting derivation to benzo(g,h,i)perylene (sediment) benchmarks for wadeable streams in NE Illinois including scatter plots of benzo(g,h,i)perylene vs. benzo(g,h,i)perylene sensitive macroinvertebrate taxa (top left), mIBI vs. benzo(g,h,i)perylene -sensitive macroinvertebrate taxa (bottom left), benzo(g,h,i)perylene vs. mIBI (top right) and a probability plot of benzo(g,h,i)perylene values by narrative ranges of the mIBI. Various benchmarks are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*



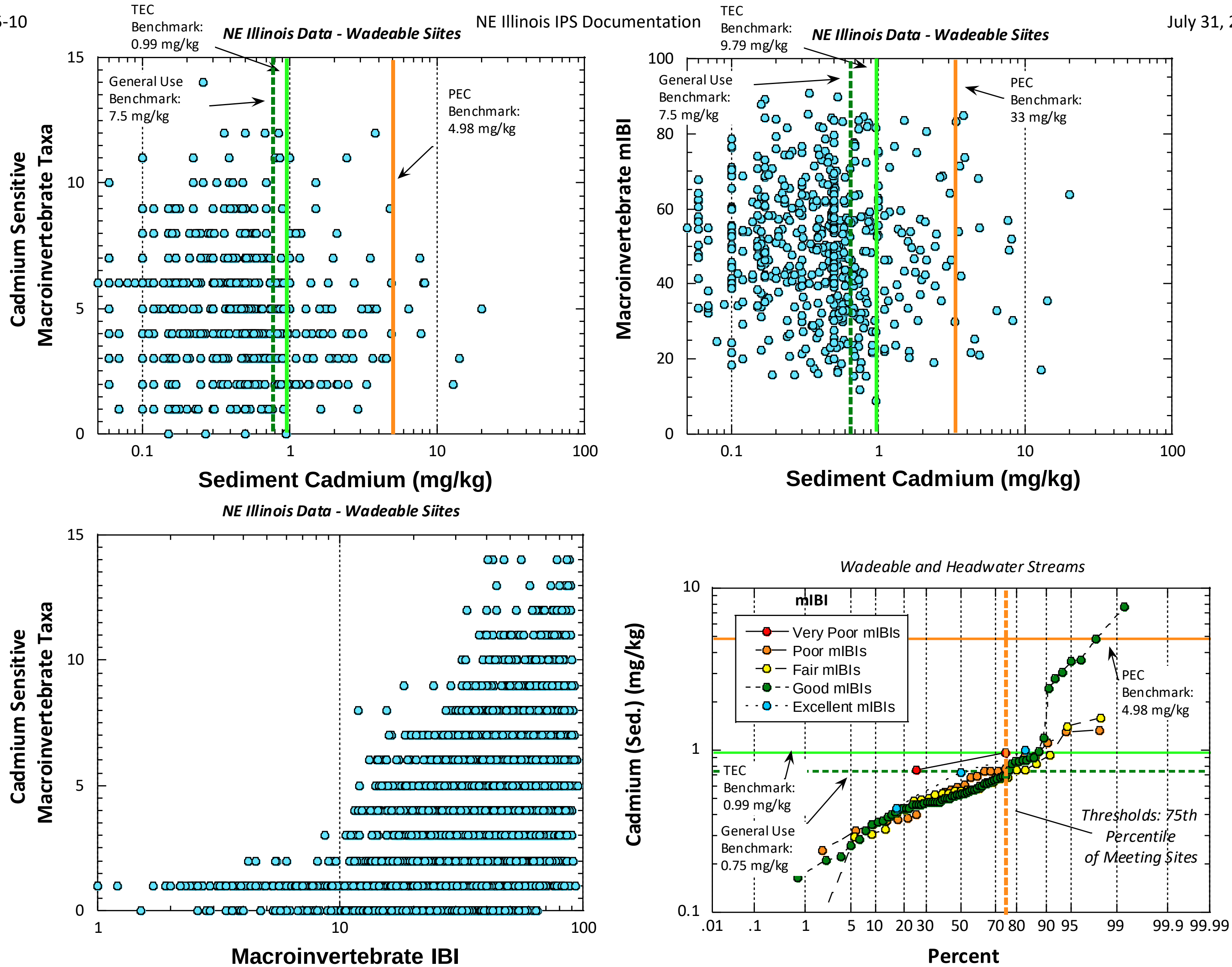

***Appendix Figure B-46.*** *Plots supporting derivation to benzo[a]anthracene (sediment) benchmarks for wadeable streams in NE Illinois including scatter plots of benzo[a]anthracene vs. benzo[a]anthracene sensitive macroinvertebrate taxa (top left), mIBI vs. benzo[a]anthracene - sensitive macroinvertebrate taxa (bottom left), benzo[a]anthracene vs. mIBI (top right) and a probability plot of benzo[a]anthracene values by narrative ranges of the mIBI. Various benchmarks are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*



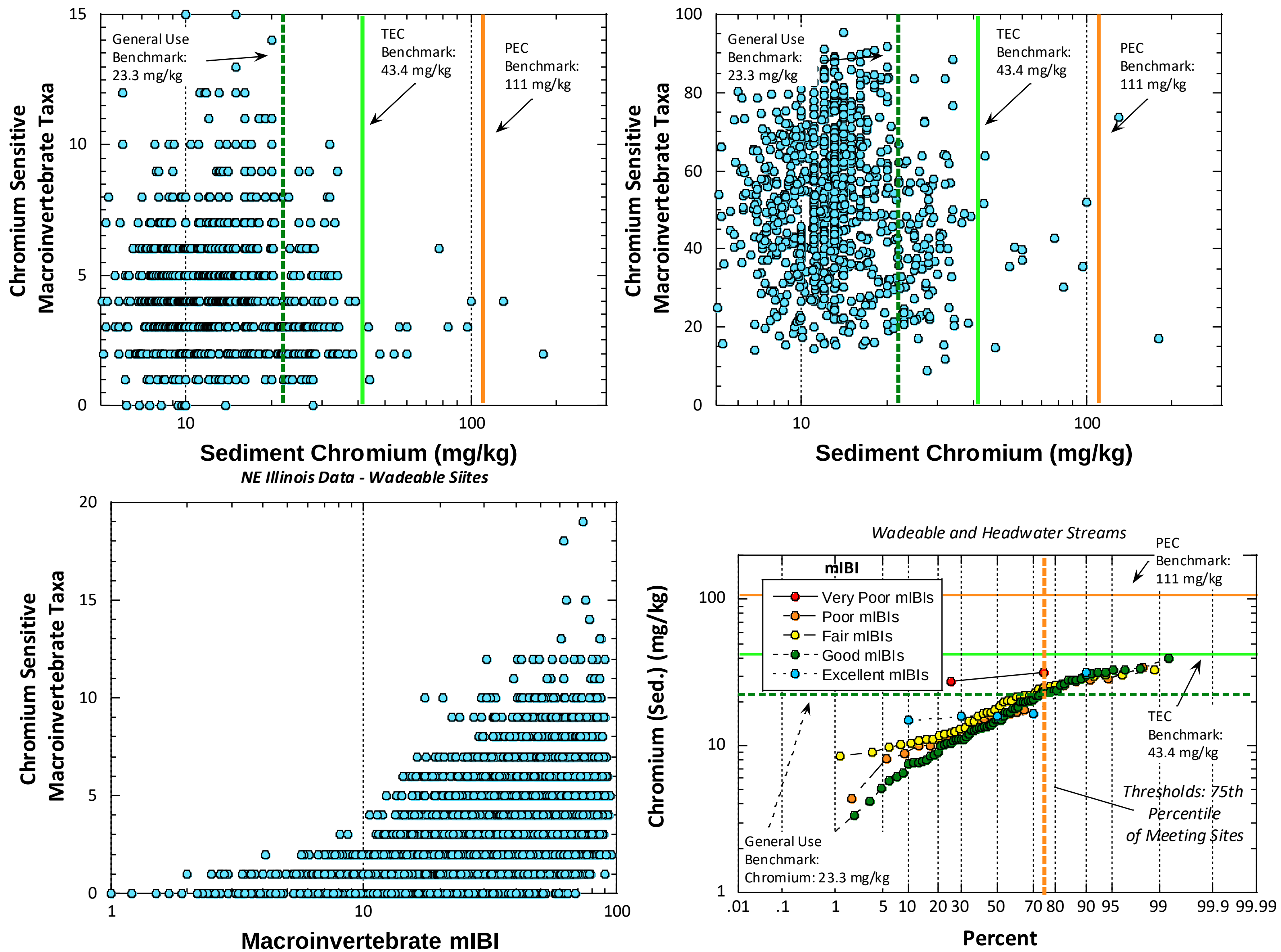

***Appendix Figure B-47.*** *Plots supporting derivation to dibenz(a,h)anthracene (sediment) benchmarks for wadeable streams in NE Illinois including scatter plots of dibenz(a,h)anthracene vs. dibenz(a,h)anthracene sensitive macroinvertebrate taxa (top left), mIBI vs. dibenz(a,h)anthracene -sensitive macroinvertebrate taxa (bottom left), dibenz(a,h)anthracene vs. mIBI (top right) and a probability plot of dibenz(a,h)anthracene values by narrative ranges of the mIBI. Various benchmarks are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*



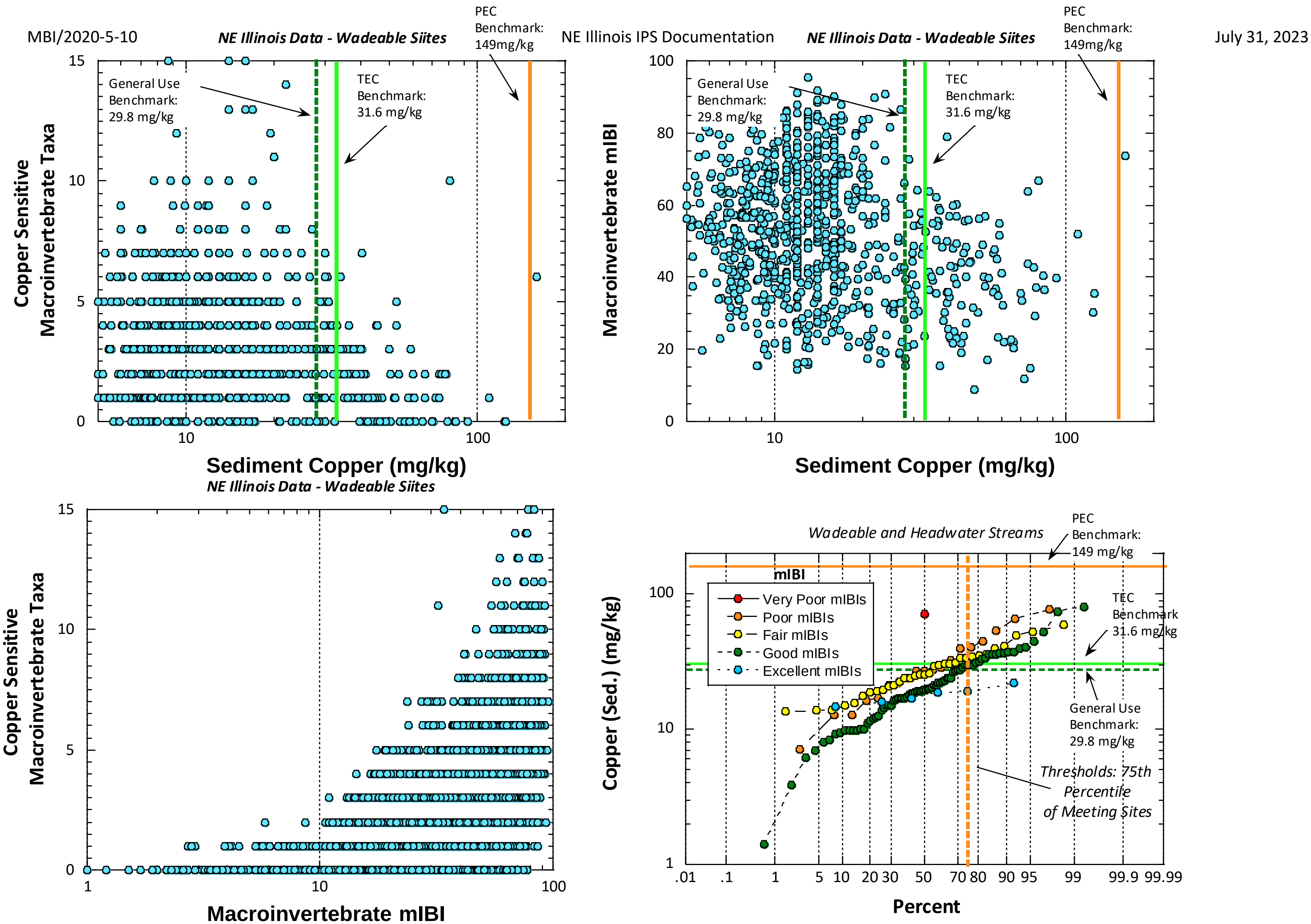

***Appendix Figure B-48.*** *Plots supporting derivation to arsenic (sediment) benchmarks for wadeable streams in NE Illinois including scatter plots of arsenic vs. arsenic sensitive macroinvertebrate taxa (top left), mIBI vs. arsenic -sensitive macroinvertebrate taxa (bottom left), arsenic vs. mIBI (top right) and a probability plot of arsenic values by narrative ranges of the mIBI. Various benchmarks are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*



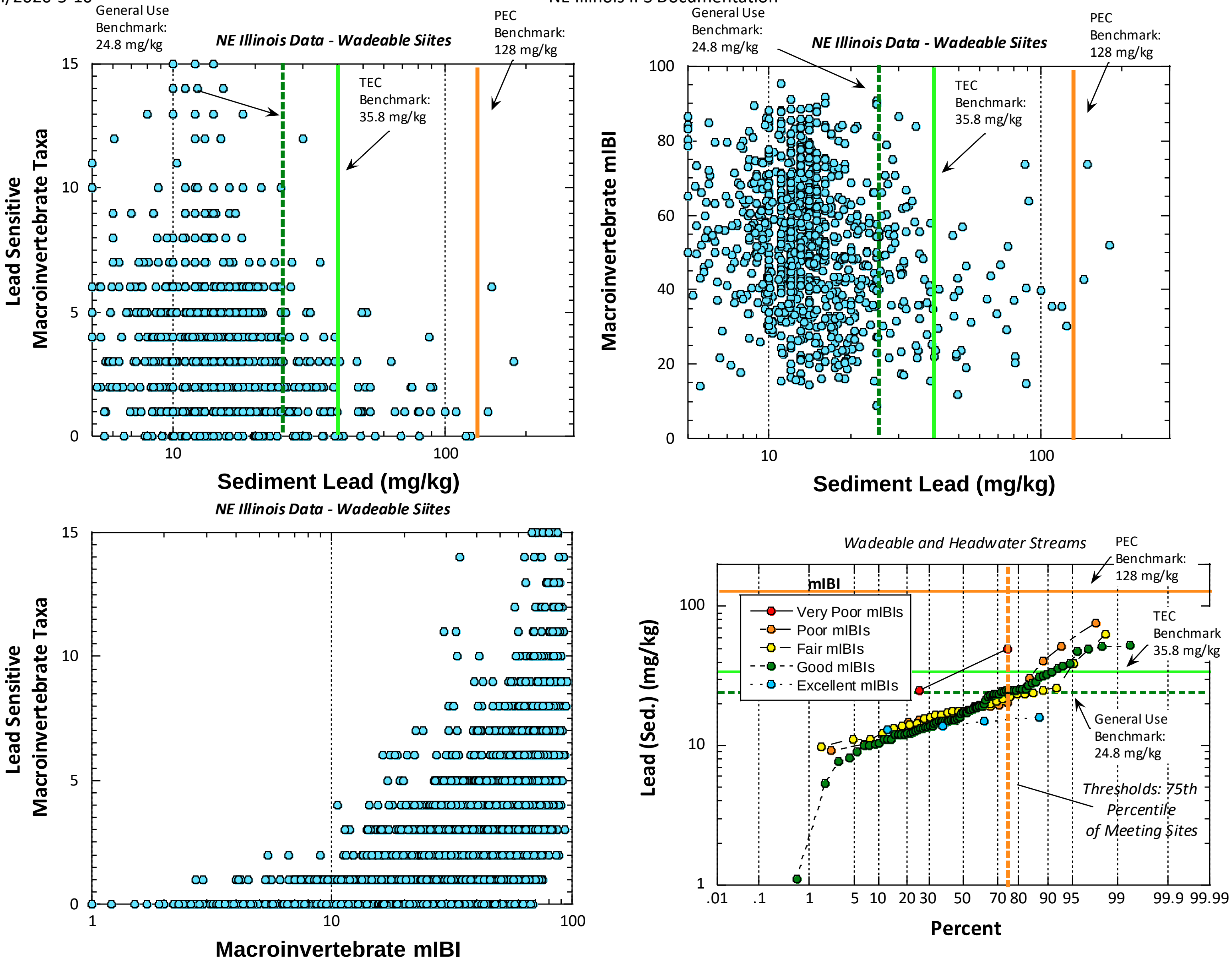

***Appendix Figure B-49.*** *Plots supporting derivation to cadmium (sediment) benchmarks for wadeable streams in NE Illinois including scatter plots of cadmium vs. cadmium sensitive macroinvertebrate taxa (top left), mIBI vs. cadmium -sensitive macroinvertebrate taxa (bottom left), cadmium vs. mIBI (top right) and a probability plot of cadmium values by narrative ranges of the mIBI. Various benchmarks are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*



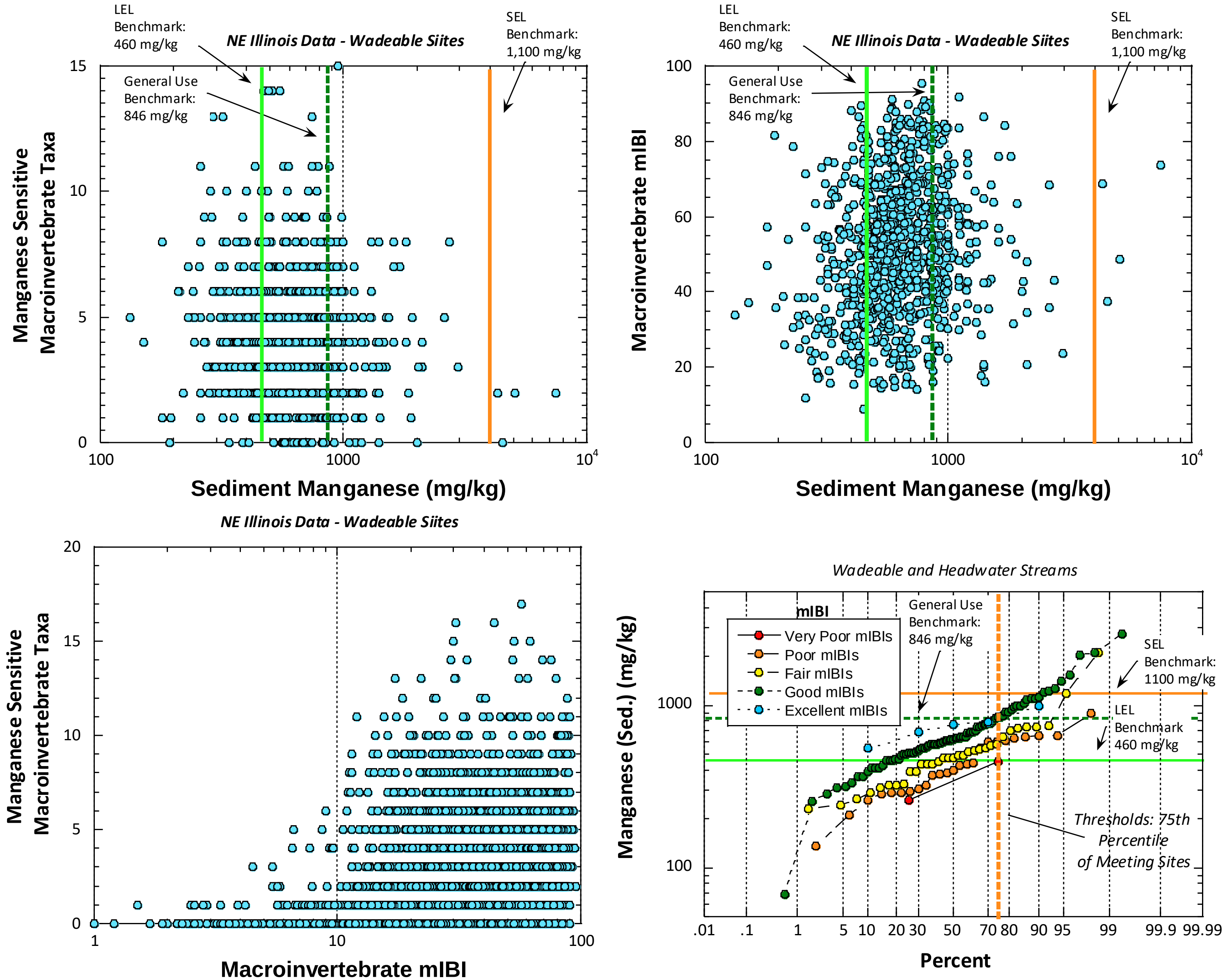

***Appendix Figure B-50.*** *Plots supporting derivation to chromium (sediment) benchmarks for wadeable streams in NE Illinois including scatter plots of chromium vs. chromium sensitive macroinvertebrate taxa (top left), mIBI vs. chromium -sensitive macroinvertebrate taxa (bottom left), chromium vs. mIBI (top right) and a probability plot of chromium values by narrative ranges of the mIBI. Various benchmarks are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*



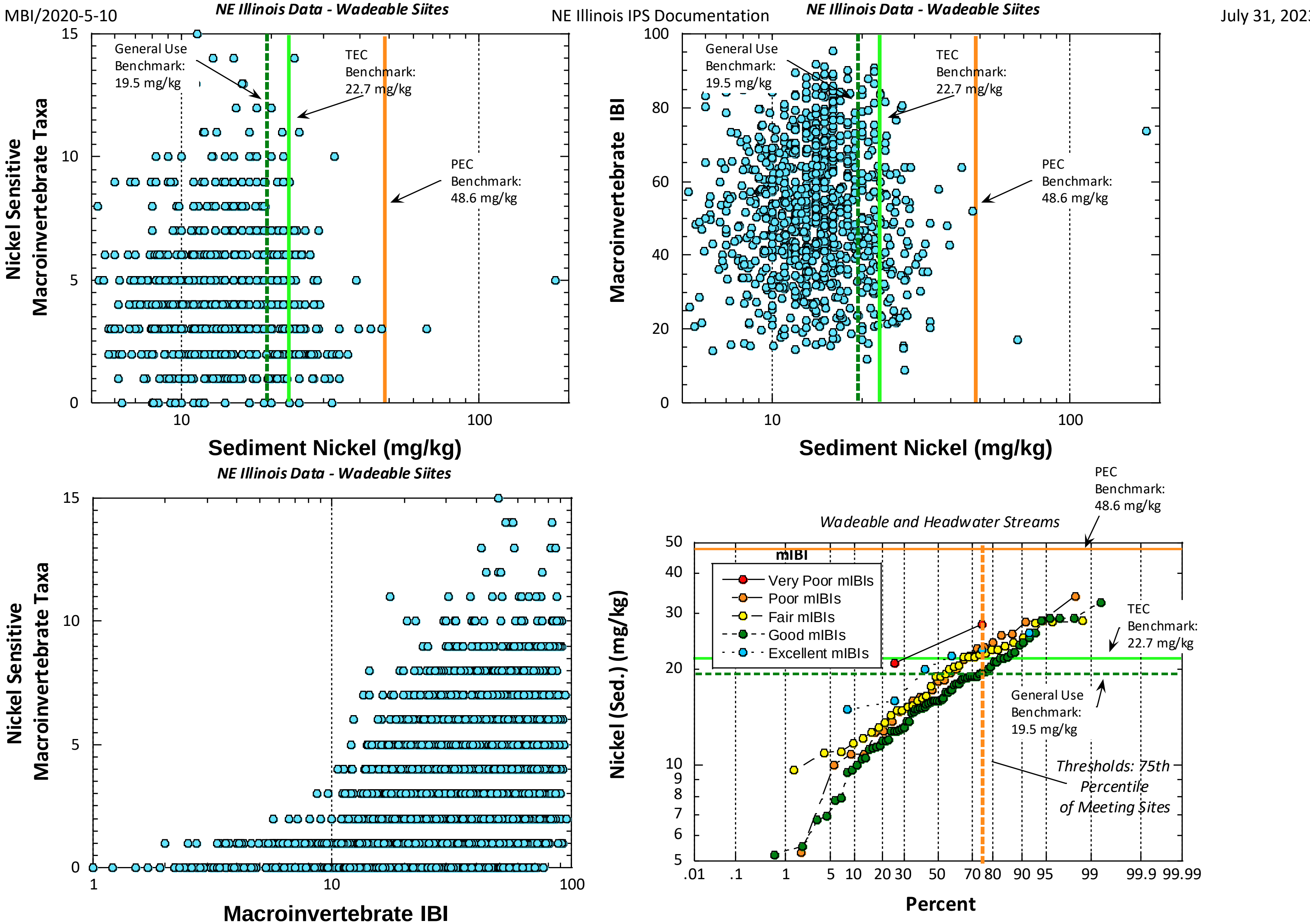

***Appendix Figure B-51.*** *Plots supporting derivation to copper (sediment) benchmarks for wadeable streams in NE Illinois including scatter plots of copper vs. copper sensitive macroinvertebrate taxa (top left), mIBI vs. copper -sensitive macroinvertebrate taxa (bottom left), copper vs. mIBI (top right) and a probability plot of copper values by narrative ranges of the mIBI. Various benchmarks are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*



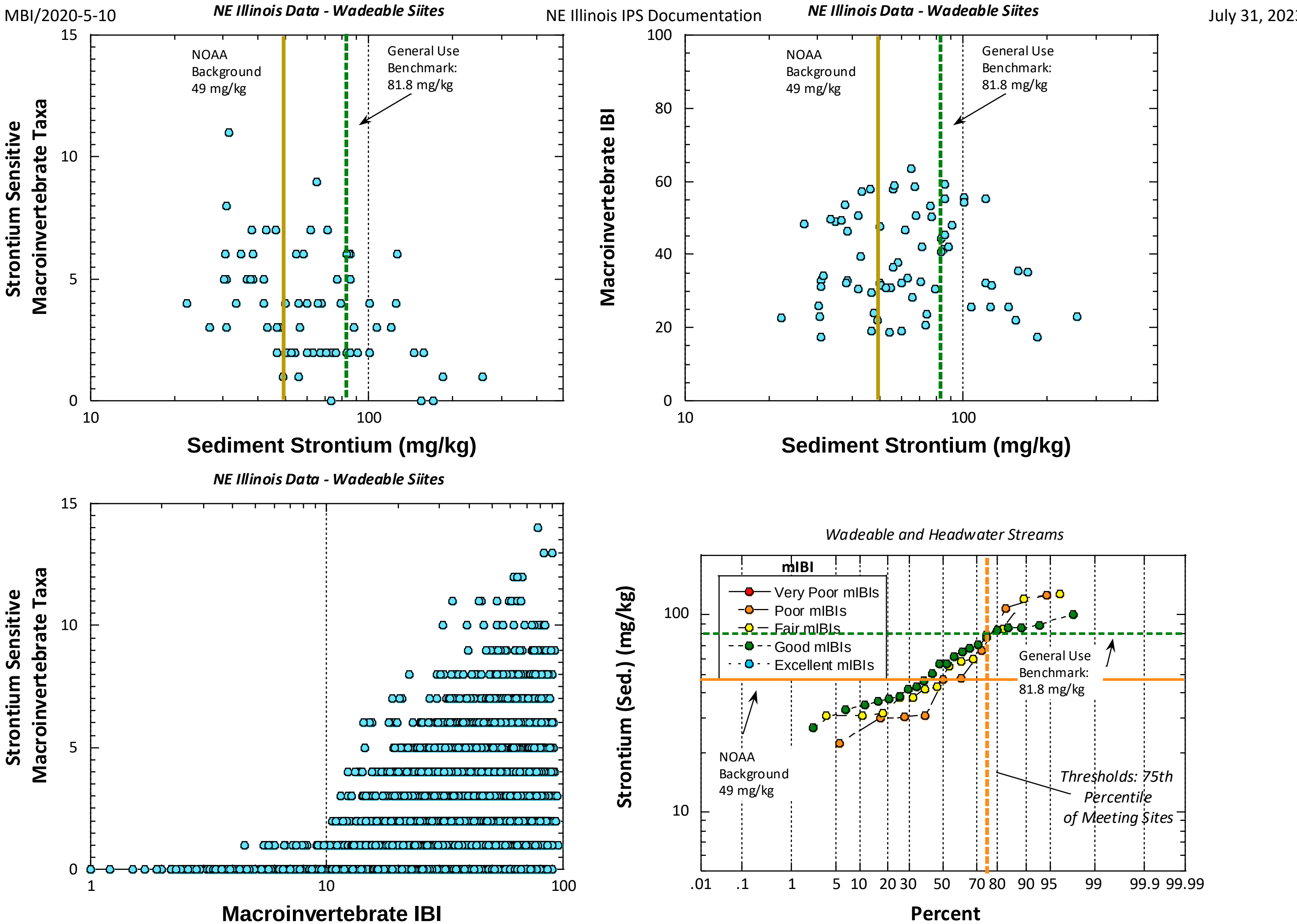

***Appendix Figure B-52.*** *Plots supporting derivation to lead (sediment) benchmarks for wadeable streams in NE Illinois including scatter plots of lead vs. lead sensitive macroinvertebrate taxa (top left), mIBI vs. lead -sensitive macroinvertebrate taxa (bottom left), lead vs. mIBI (top right) and a probability plot of lead values by narrative ranges of the mIBI. Various benchmarks are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*



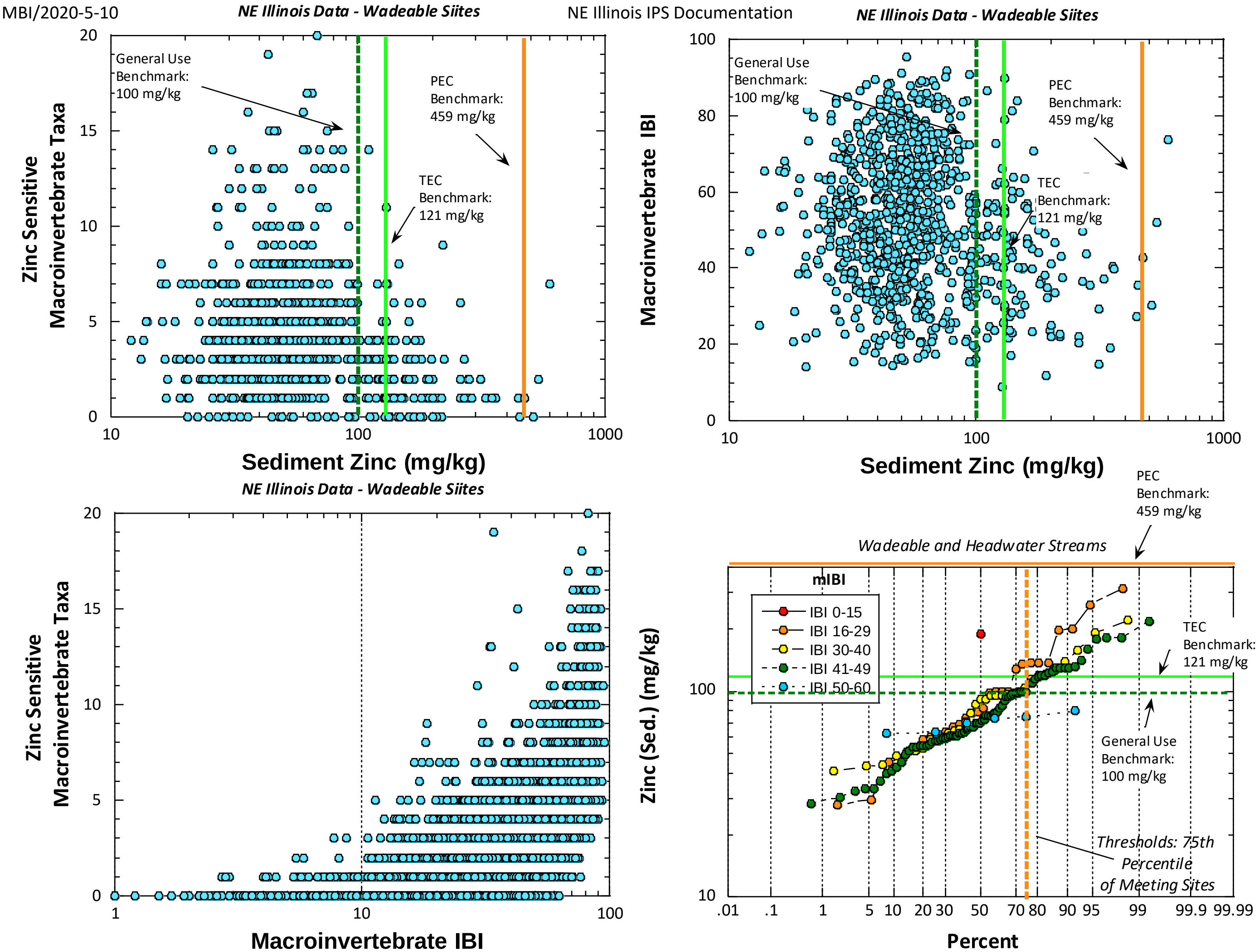

***Appendix Figure B-53.*** *Plots supporting derivation to manganese (sediment) benchmarks for wadeable streams in NE Illinois including scatter plots of manganese vs. manganese sensitive macroinvertebrate taxa (top left), mIBI vs. manganese -sensitive macroinvertebrate taxa (bottom left), manganese vs. mIBI (top right) and a probability plot of manganese values by narrative ranges of the mIBI. Various benchmarks are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*



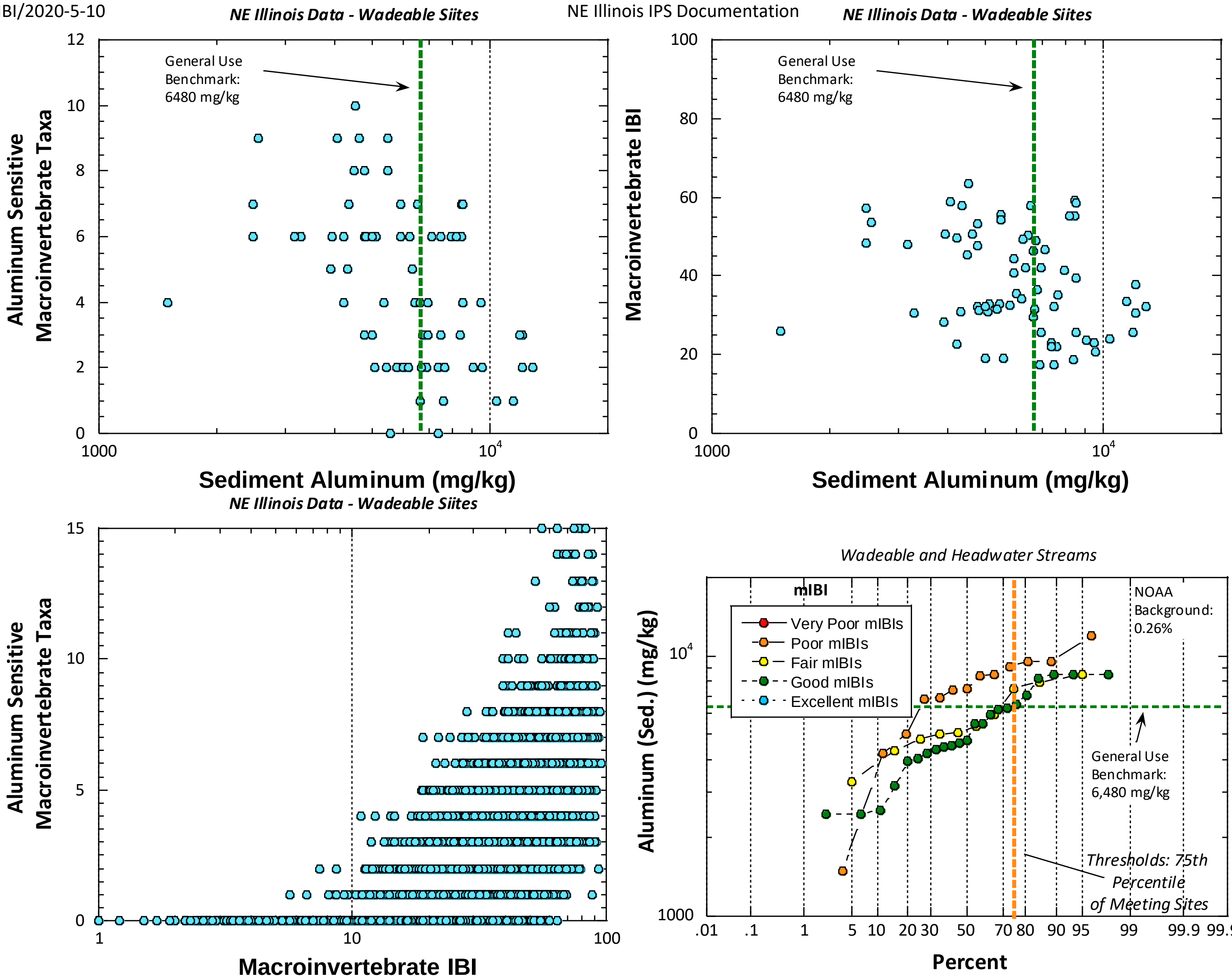

***Appendix Figure B-54.*** *Plots supporting derivation to nickel (sediment) benchmarks for wadeable streams in NE Illinois including scatter plots of nickel vs. nickel sensitive macroinvertebrate taxa (top left), mIBI vs. nickel -sensitive macroinvertebrate taxa (bottom left), nickel vs. mIBI (top right) and a probability plot of nickel values by narrative ranges of the mIBI. Various benchmarks are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*



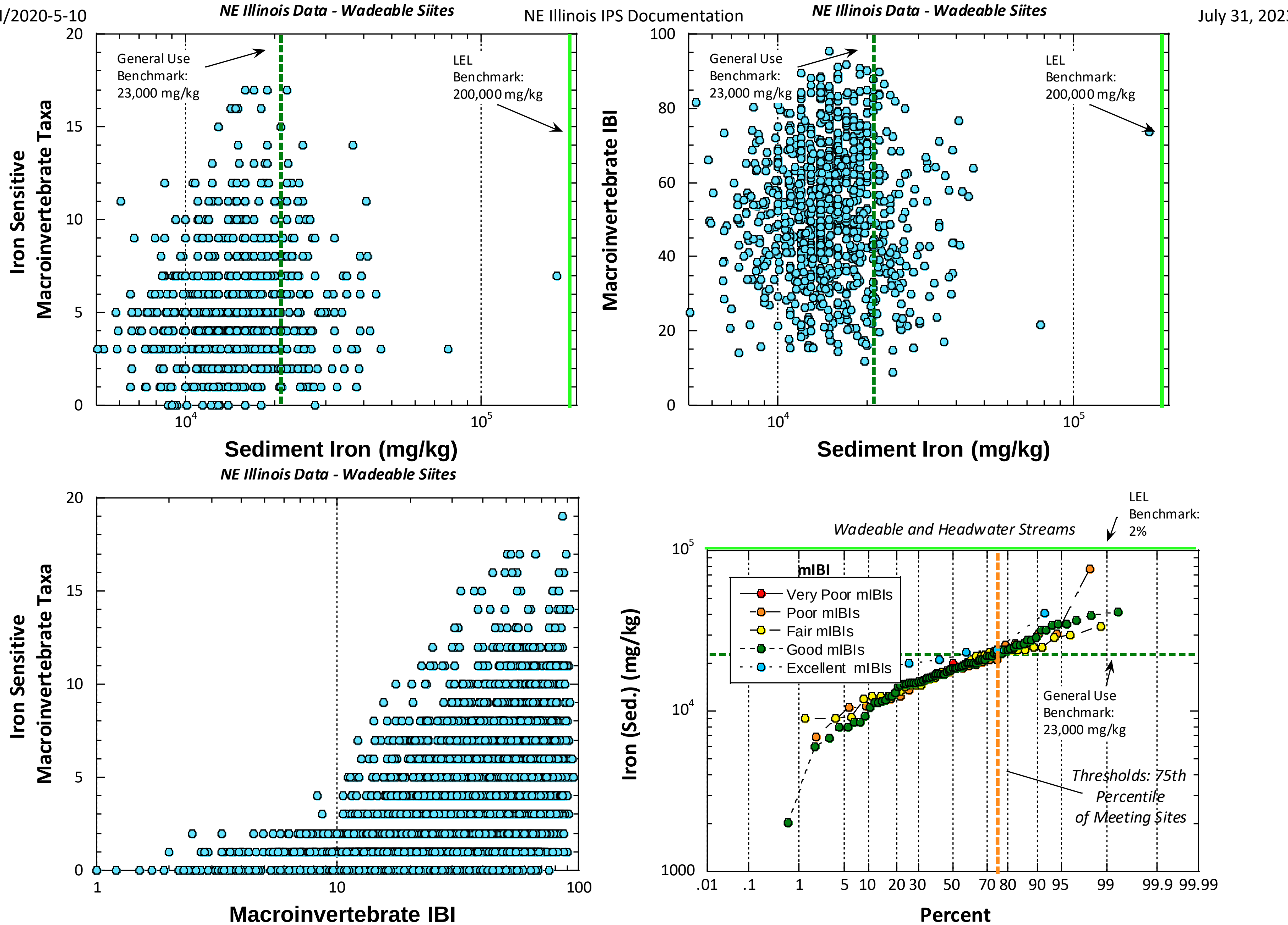

***Appendix Figure 55.*** *Plots supporting derivation to strontium (sediment) benchmarks for wadeable streams in NE Illinois including scatter plots of strontium vs. strontium sensitive macroinvertebrate taxa (top left), mIBI vs. strontium -sensitive macroinvertebrate taxa (bottom left), strontium vs. mIBI (top right) and a probability plot of strontium values by narrative ranges of the mIBI. Various benchmarks are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*




***Appendix Figure 56.*** *Plots supporting derivation to zinc (sediment) benchmarks for wadeable streams in NE Illinois including scatter plots of zinc vs. zinc sensitive macroinvertebrate taxa (top left), mIBI vs. zinc -sensitive macroinvertebrate taxa (bottom left), zinc vs. mIBI (top right) and a probability plot of zinc values by narrative ranges of the mIBI. Various benchmarks are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*




***Appendix Figure B-57.*** *Plots supporting derivation of aluminum(sediment) benchmarks for wadeable streams in NE Illinois including scatter plots of aluminum vs. aluminum sensitive macroinvertebrate taxa (top left), mIBI vs. aluminum -sensitive macroinvertebrate taxa (bottom left), aluminum vs. mIBI (top right) and a probability plot of aluminum values by narrative ranges of the mIBI. Various benchmarks are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*

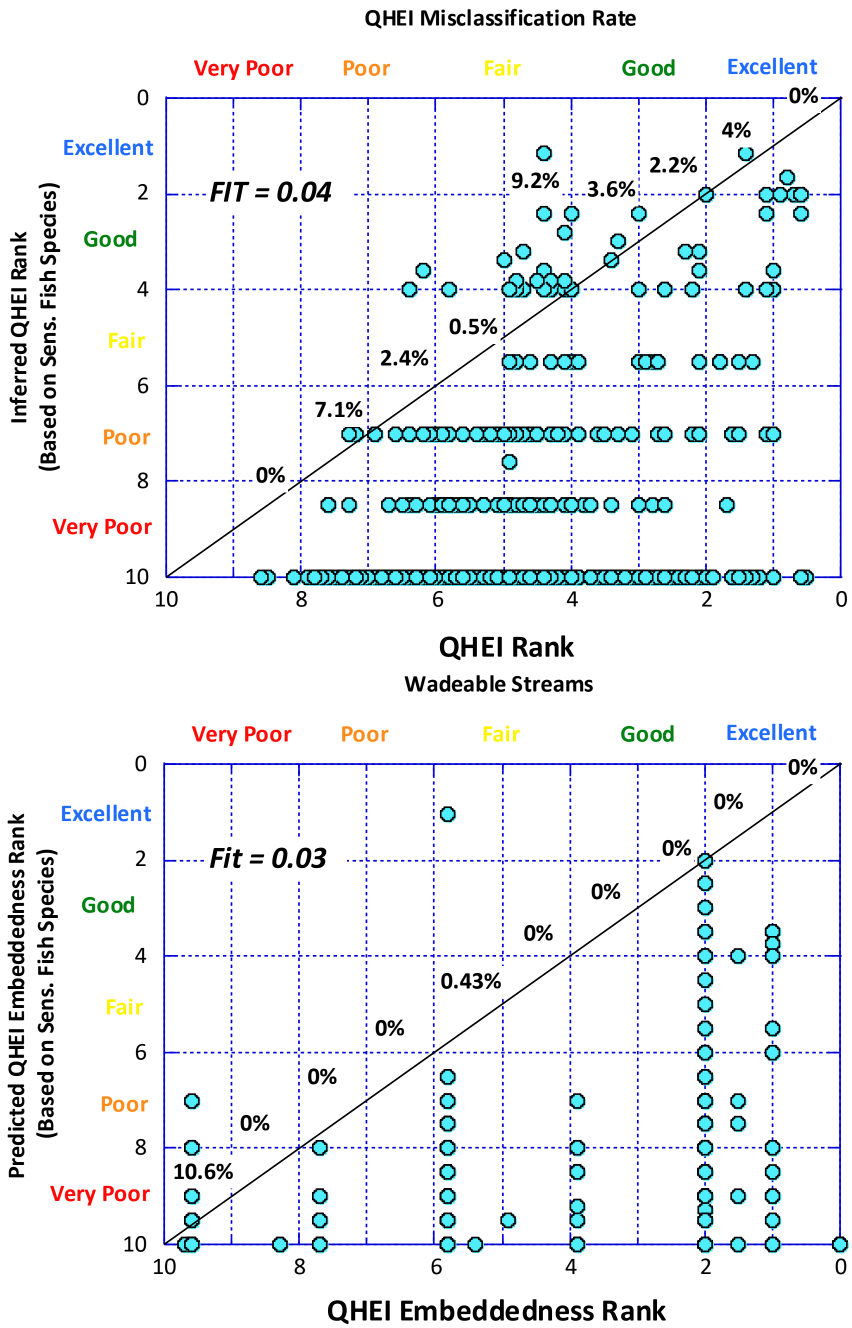



***Appendix Figure B-58.*** *Plots supporting derivation of iron(sediment) thresholds for wadeable streams in NE Illinois including scatter plots of iron vs. iron sensitive macroinvertebrate taxa (top left), mIBI vs. iron -sensitive macroinvertebrate taxa (bottom left), iron vs. mIBI (top right) and a probability plot of iron values by narrative ranges of the mIBI. Various thresholds are represented by horizontal or vertical lines. Data from IL IPS study area sites in NE Illinois (see text).*

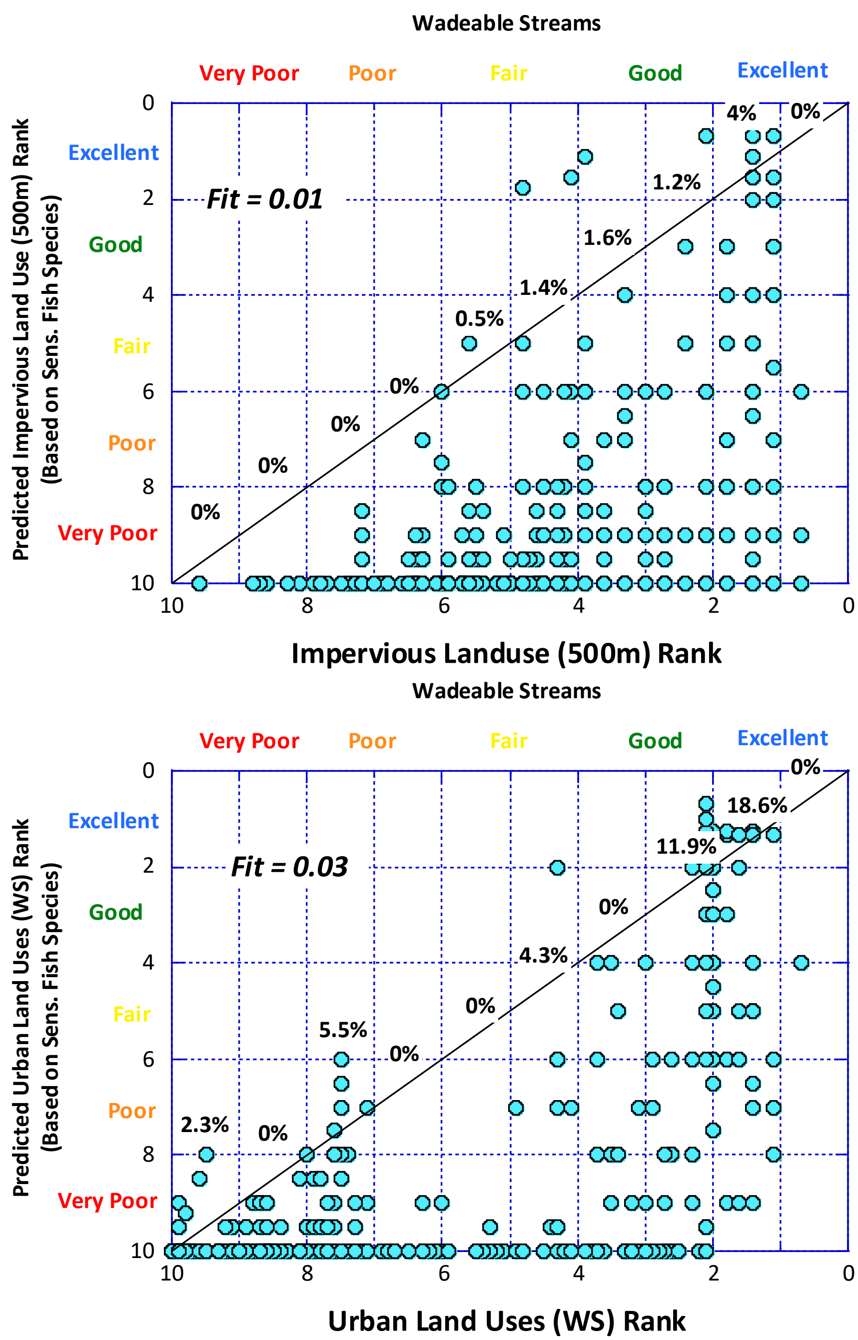
# Appendix C: Selected plots of ambient stressor ranks for key IPS stressors vs. predicted stressor ranks based stressor-specific sensitive fish species or macroinvertebrate taxa for headwater and wadeable stream sites in the IPS study area.

## Appendix C Description

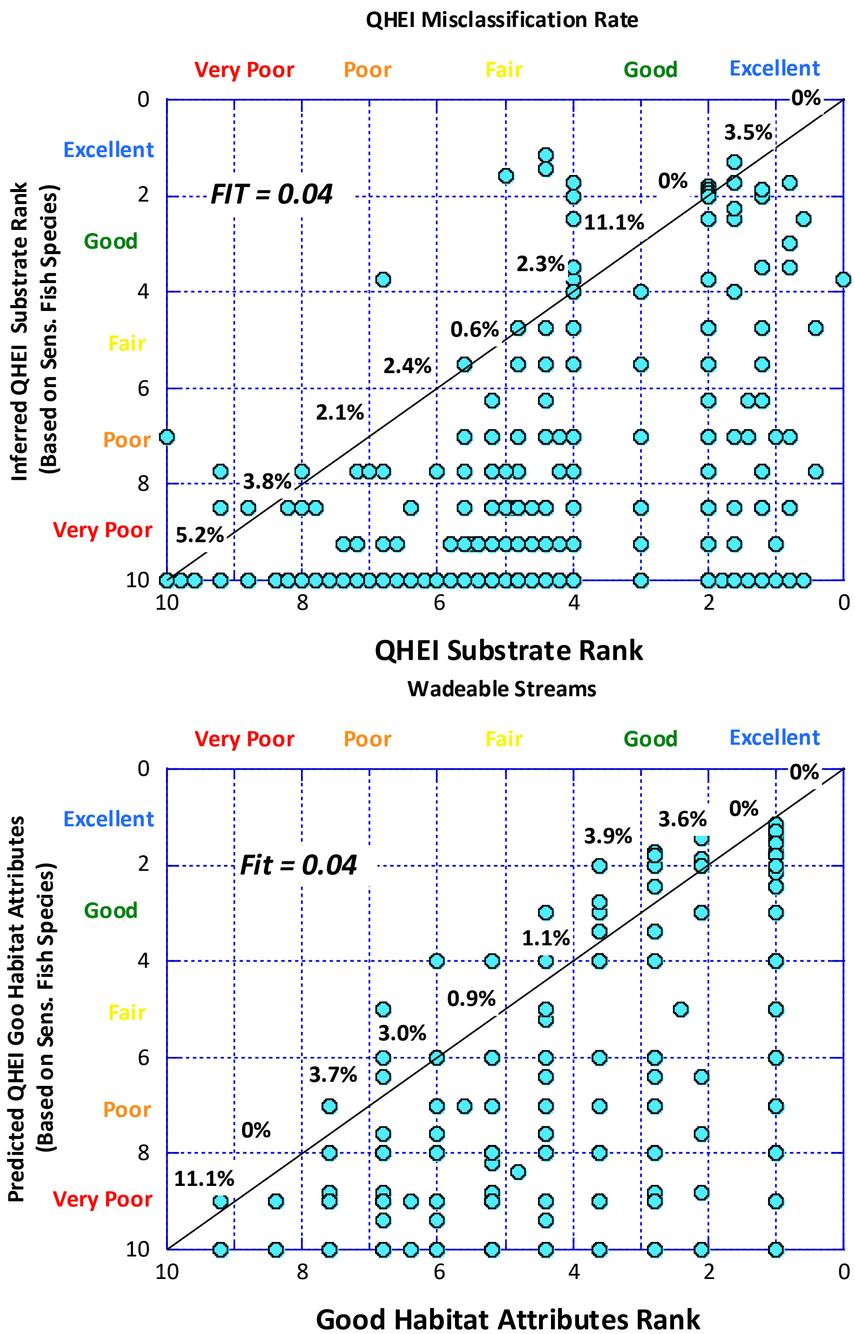
The intercept line reflects a presumed 1:1 prediction with ambient stressor ranks. A percent prediction "error" is determined where more sensitive fish species or macroinvertebrate taxa occur than are expected based on stressor levels for integer levels of stressor effects that are presented as percentages. The FIT coefficient is a measure that is inverse to the percentage of "errors" and the magnitude of those errors represented by the magnitude of deviation from the expected sensitive fish species or macroinvertebrate taxa. Presumably, a lower FIT score represents a better stress:response relationship for a particular stressor. This appendix includes a representative selection from different stressor categories. Please contact MBI for plots of stressors not included herein.



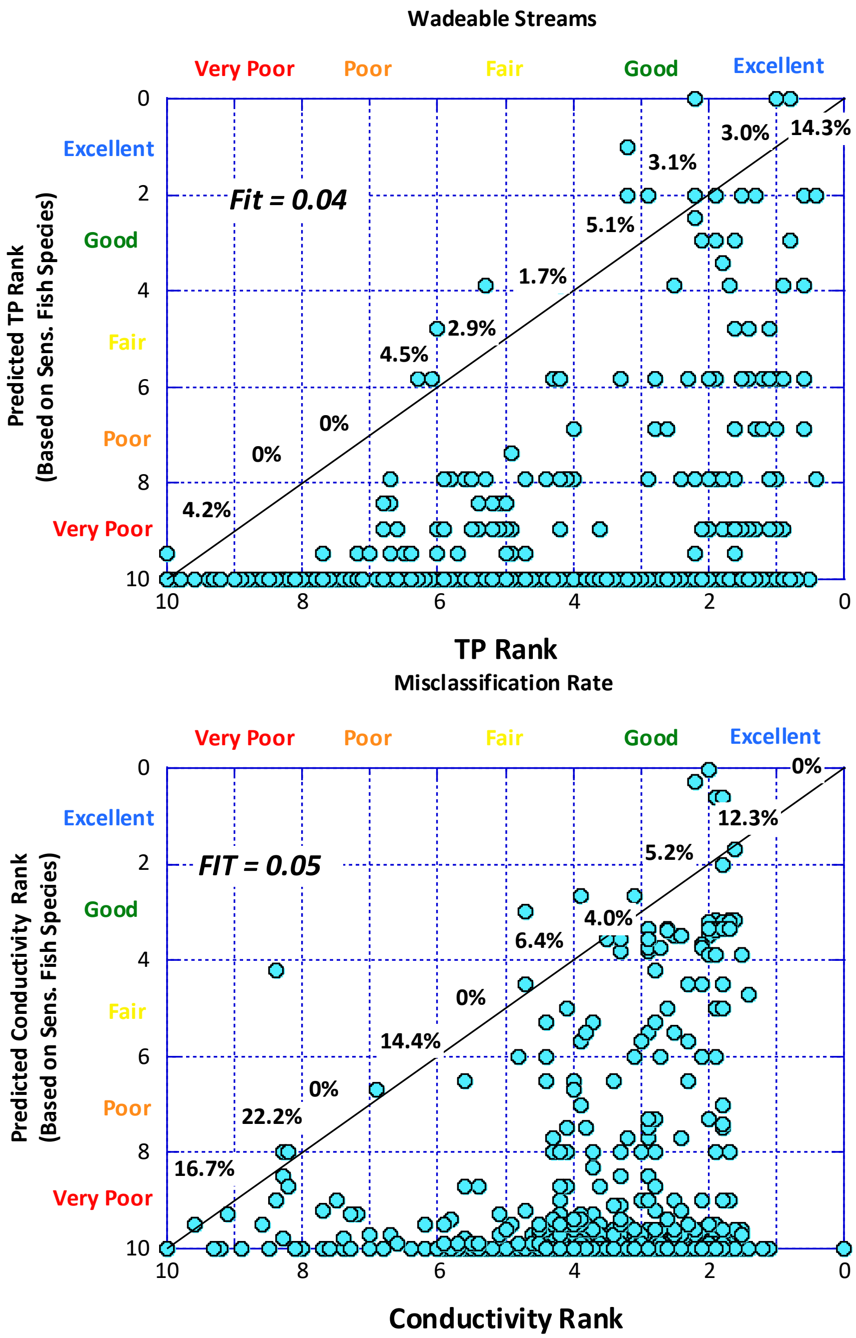

***Appendix Figure C-1***. *Plots of QHEI overall (top) and QHEI embeddedness (bottom) stressor rank vs. predicted stressor ranks based stressor-specific sensitive fish species for headwater and wadeable stream sites in the IPS study area. Line reflects 1:1 prediction with ambient stressor ranks. Prediction "error" where more sensitive fish species occur than expected based on stressor level for integer levels of stressor effects are presented as percentages. FIT coefficient is a measure that increases with the percent of errors and the magnitude of errors (i.e., magnitude of deviation from expected sensitive fish species).*



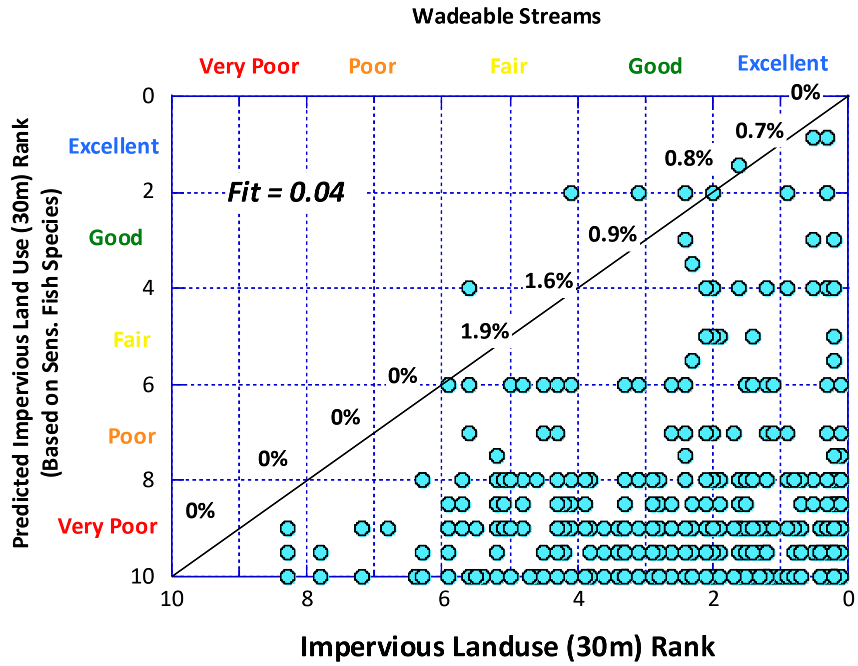

***Appendix Figure C-2**. Plots of Impervious Land Use (500m buffer, top) and Urban Land Uses (watershed scale, bottom) vs. predicted stressor ranks based stressor-specific sensitive fish species for headwater and wadeable stream sites in the IPS study area. Line reflects 1:1 prediction with ambient stressor ranks. Prediction "error" where more sensitive fish species occur than expected based on stressor level for integer levels of stressor effects are presented as percentages. FIT coefficient is a measure that increases with the percent of errors and the magnitude of errors (i.e., magnitude of deviation from expected sensitive fish species).*

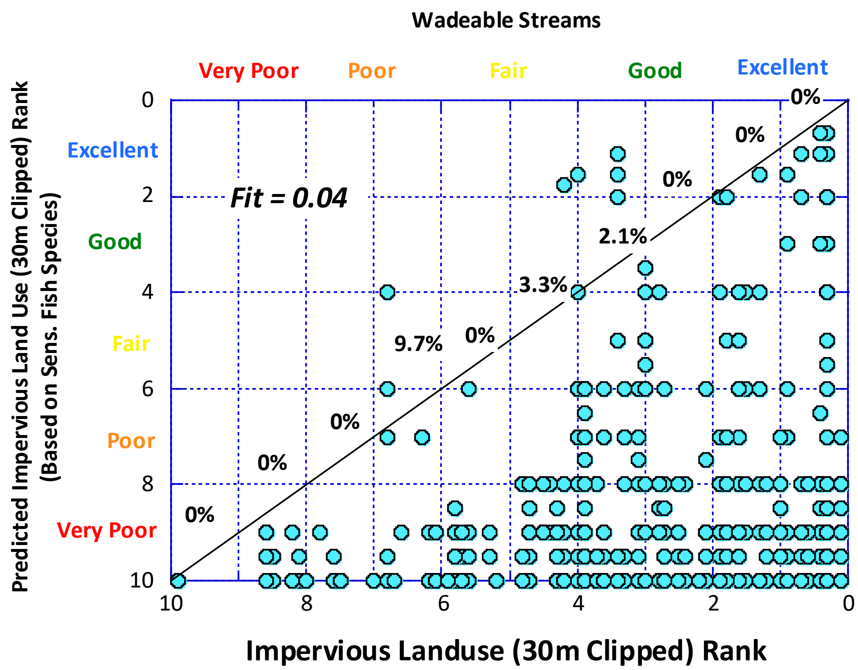



***Appendix Figure C-3****. Plots QHEI Substrate Score (top) and QHEI Good Habitat Attributes (bottom) vs. predicted stressor ranks based on stressor-specific sensitive fish species for headwater and wadeable stream sites in the IPS study area. Line reflects 1:1 prediction with ambient stressor ranks. Prediction "error" where more sensitive fish species occur than expected based on stressor level for integer levels of stressor effects are presented as percentages. FIT coefficient is a measure that increases with the percent of errors and the magnitude of errors (i.e., magnitude of deviation from expected sensitive fish species).*



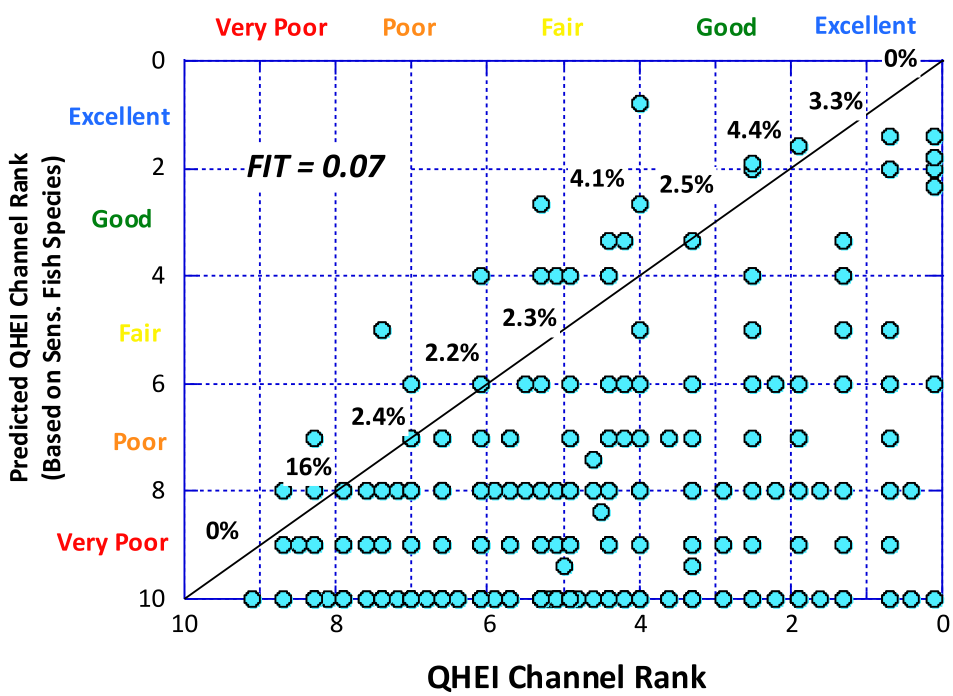

***Appendix Figure C-4***. *Plots of total phosphorus (top) and conductivity (bottom) stressor ranks vs. predicted stressor ranks based on stressor-specific sensitive fish species for headwater and wadeable stream sites in the IPS study area. Line reflects 1:1 prediction with ambient stressor ranks. Prediction "error" where more sensitive fish species occur than expected based on stressor level for integer levels of stressor effects are presented as percentages. FIT coefficient is a measure that increases with the percent of errors and the magnitude of errors (i.e., magnitude of deviation from expected sensitive fish species).*



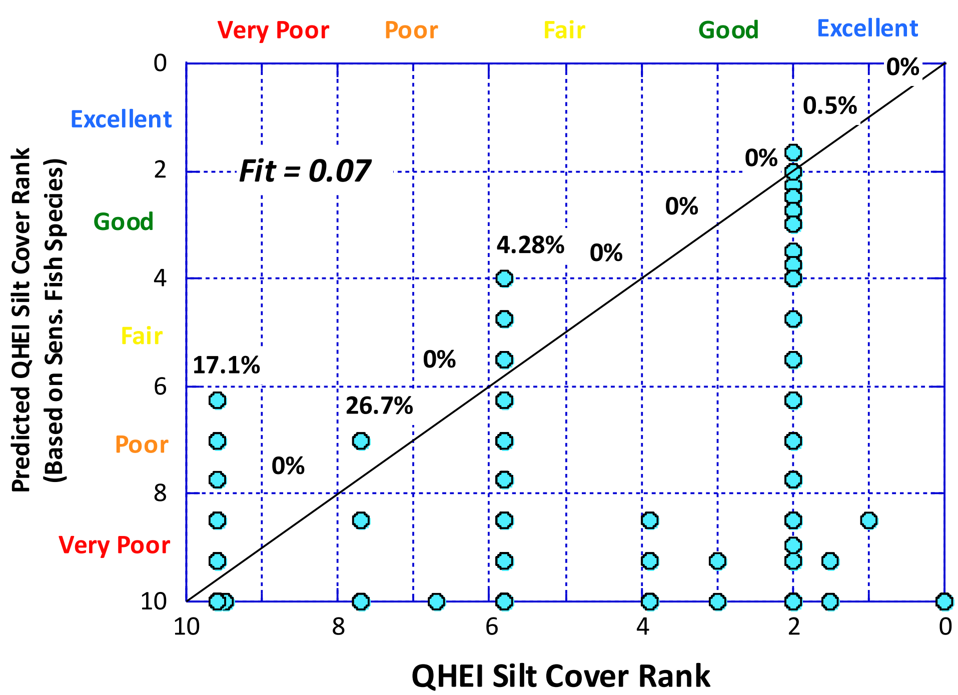




***Appendix Figure C-5***. *Plots of 30m Buffer Impervious Land Use (top) and 30m Clipped Buffer Impervious Land Use (bottom) stressor ranks vs. predicted stressor ranks based on stressor-specific sensitive fish species for headwater and wadeable stream sites in the IPS study area. Line reflects 1:1 prediction with ambient stressor ranks. Prediction "error" where more sensitive fish species occur than expected based on stressor level for integer levels of stressor effects are presented as percentages. FIT coefficient is a measure that increases with the percent of errors and the magnitude of errors (i.e., magnitude of deviation from expected sensitive fish species).*

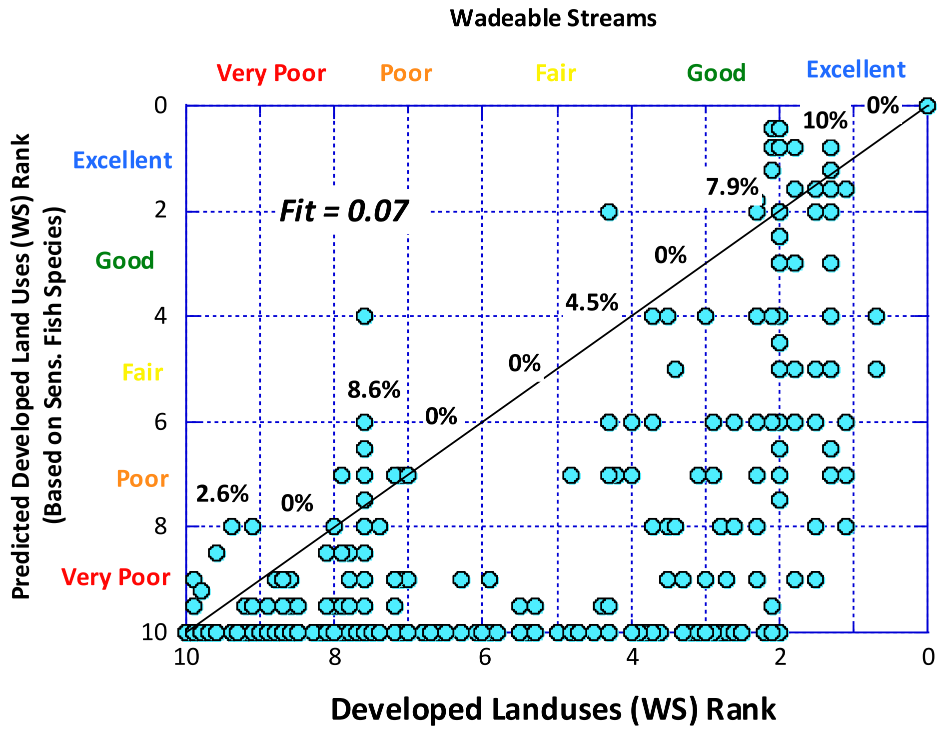
QHEI Misclassification Rate

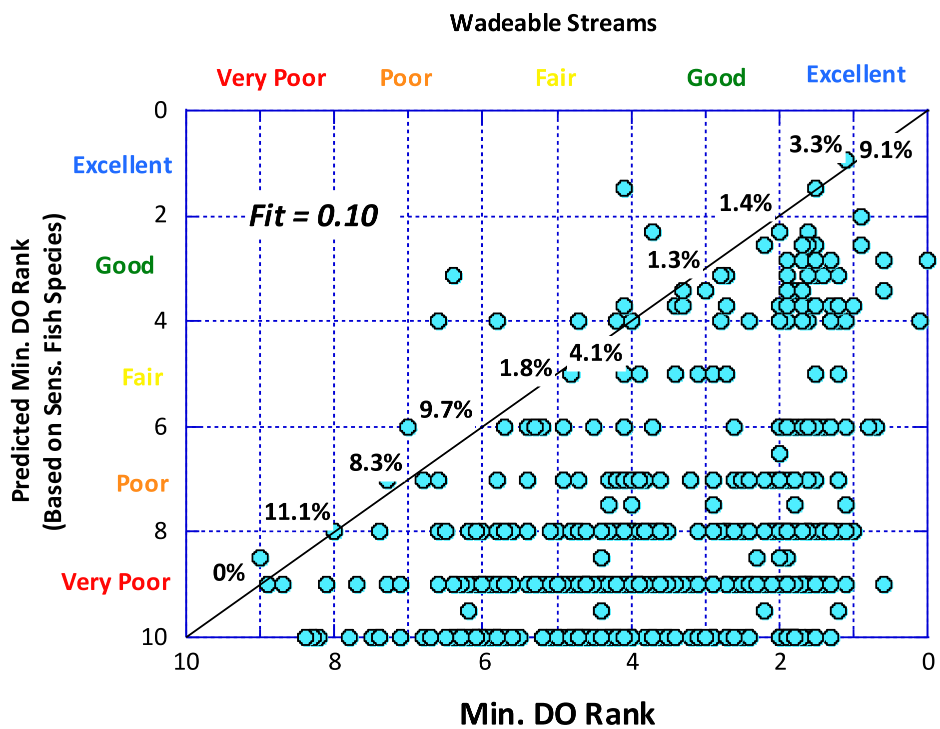



Wadeable Streams

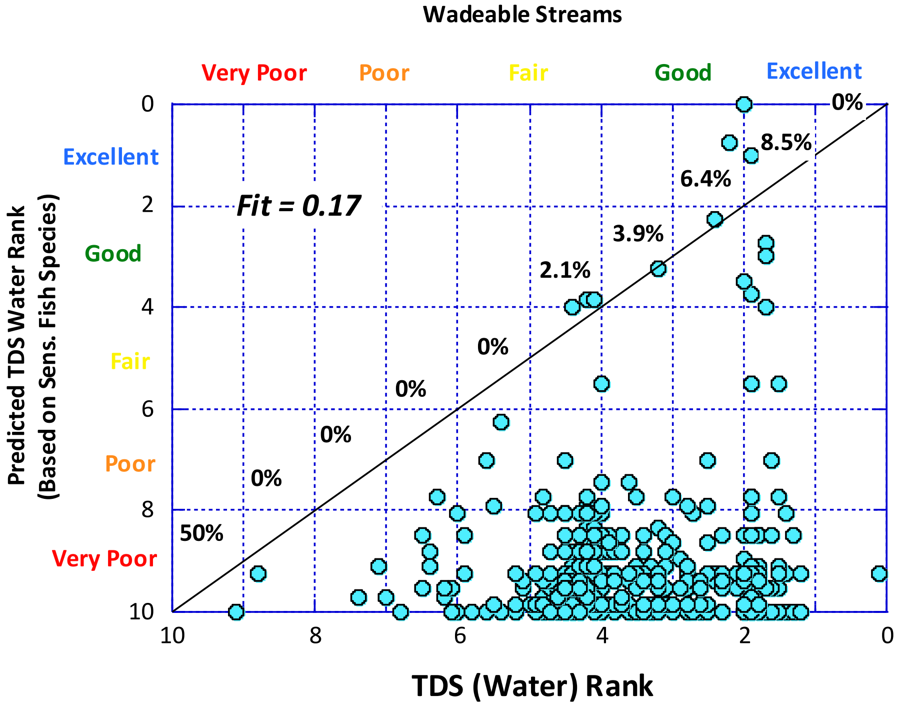


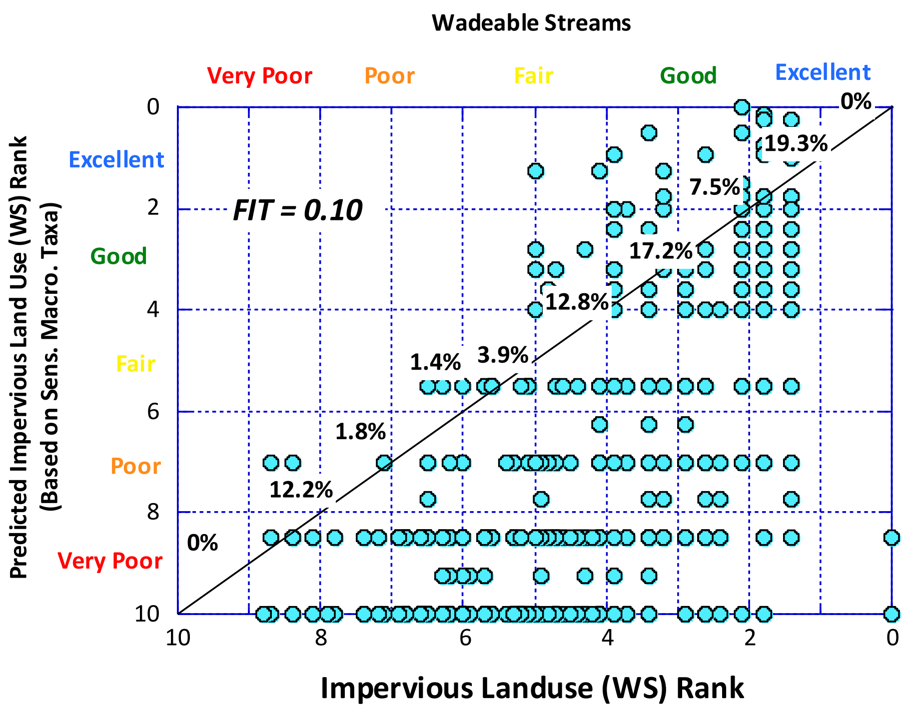

***Appendix Figure C-6****. Plots of QHEI Channel Score (top) and QHEI Silt Cover Score (bottom) ranks vs. predicted stressor ranks based on stressor-specific sensitive fish species for headwater and wadeable stream sites in the IPS study area. Line reflects 1:1 prediction with ambient stressor ranks. Prediction “error” where more sensitive fish species occur than expected based on stressor level for integer levels of stressor effects are presented as percentages. FIT coefficient is a measure that increases with the percent of errors and the magnitude of errors (i.e., magnitude of deviation from expected sensitive fish species).*



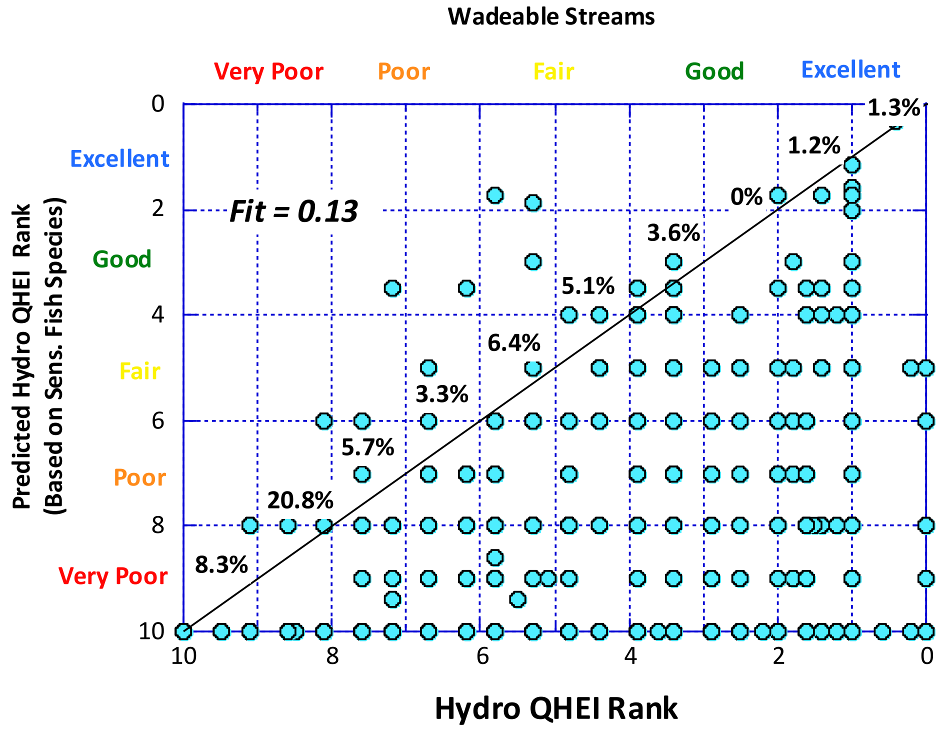



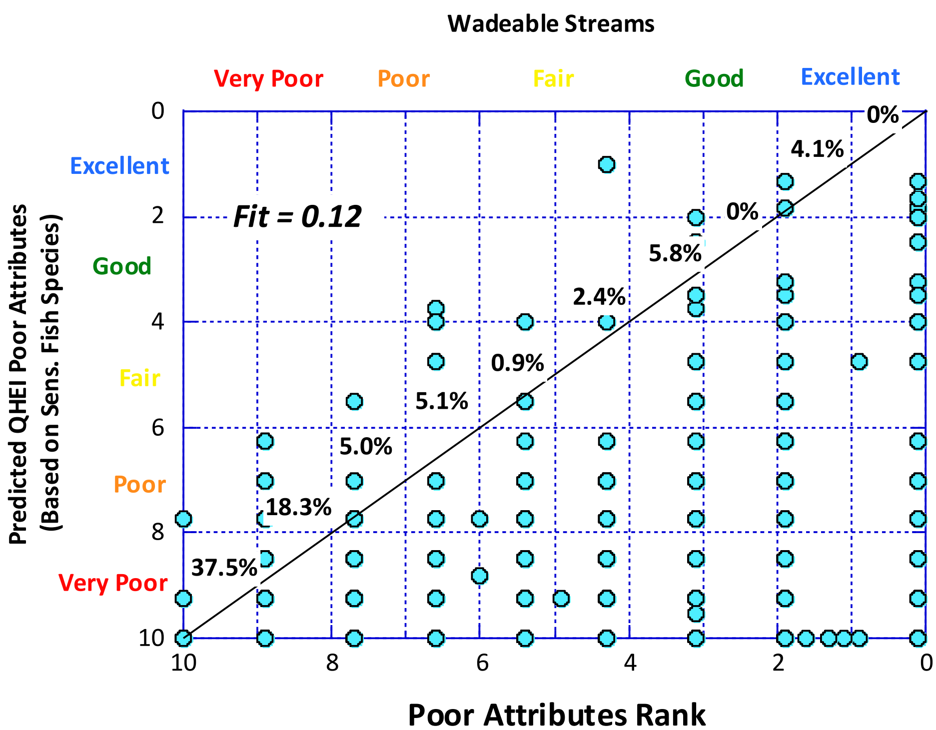

***Appendix Figure C-7***. *Plots of Developed Land Use Score (WS) (top) and Min. Dissolved Oxygen (bottom) ranks vs. predicted stressor ranks based on stressor-specific sensitive fish species for headwater and wadeable stream sites in the IPS study area. Line reflects 1:1 prediction with ambient stressor ranks. Prediction "error" where more sensitive fish species occur than expected based on stressor level for integer levels of stressor effects are presented as percentages. FIT coefficient is a measure that increases with the percent of errors and the magnitude of errors (i.e., magnitude of deviation from expected sensitive fish species).*



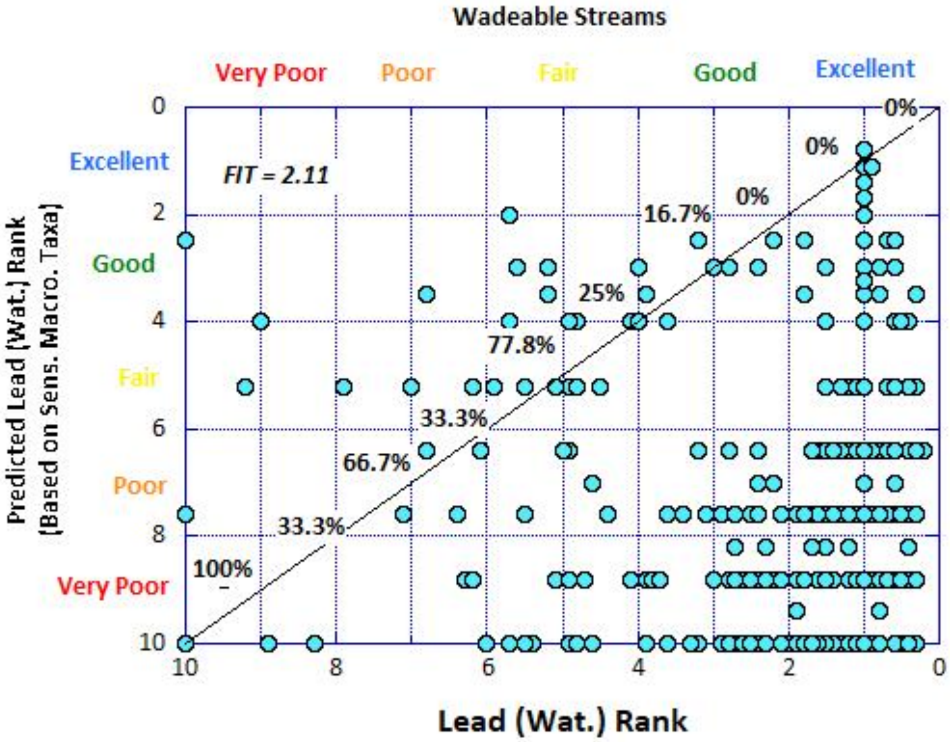



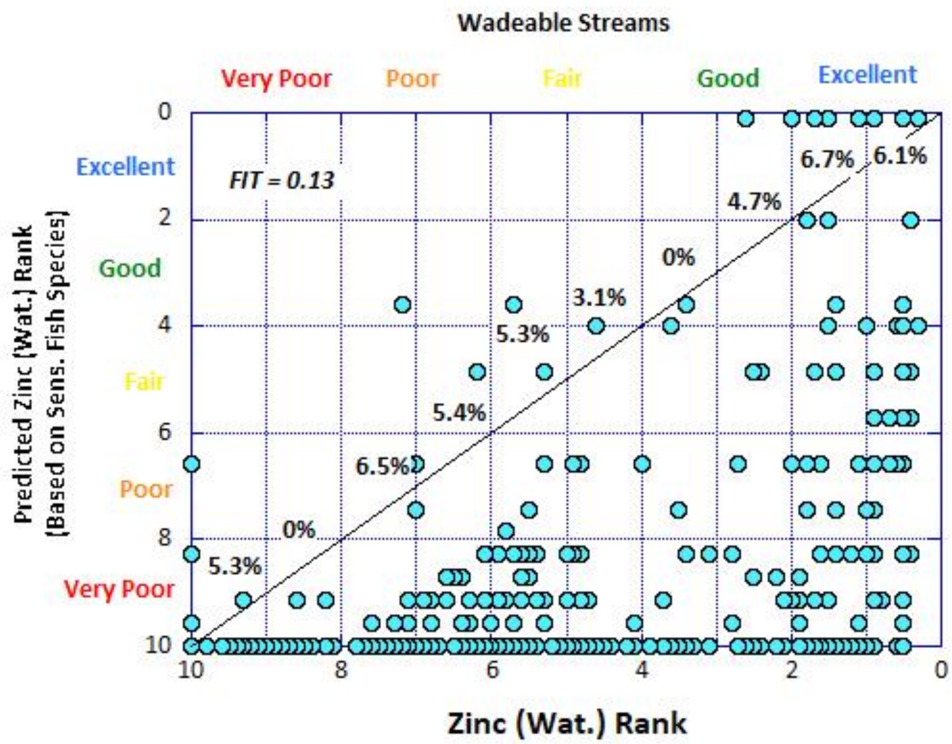

***Appendix Figure C-8****. Plots of TDS rank (top) and Impervious Land Use at the watershed scale (bottom) rank vs. predicted stressor ranks based on stressor-specific sensitive fish species (top) and macroinvertebrate taxa (bottom) for headwater and wadeable stream sites in the IPS study area. Line reflects 1:1 prediction with ambient stressor ranks. Prediction "error" where more sensitive fish species occur than expected based on stressor level for integer levels of stressor effects are presented as percentages. FIT coefficient is a measure that increases with the percent of errors and the magnitude of errors (i.e., magnitude of deviation from expected sensitive fish species).*



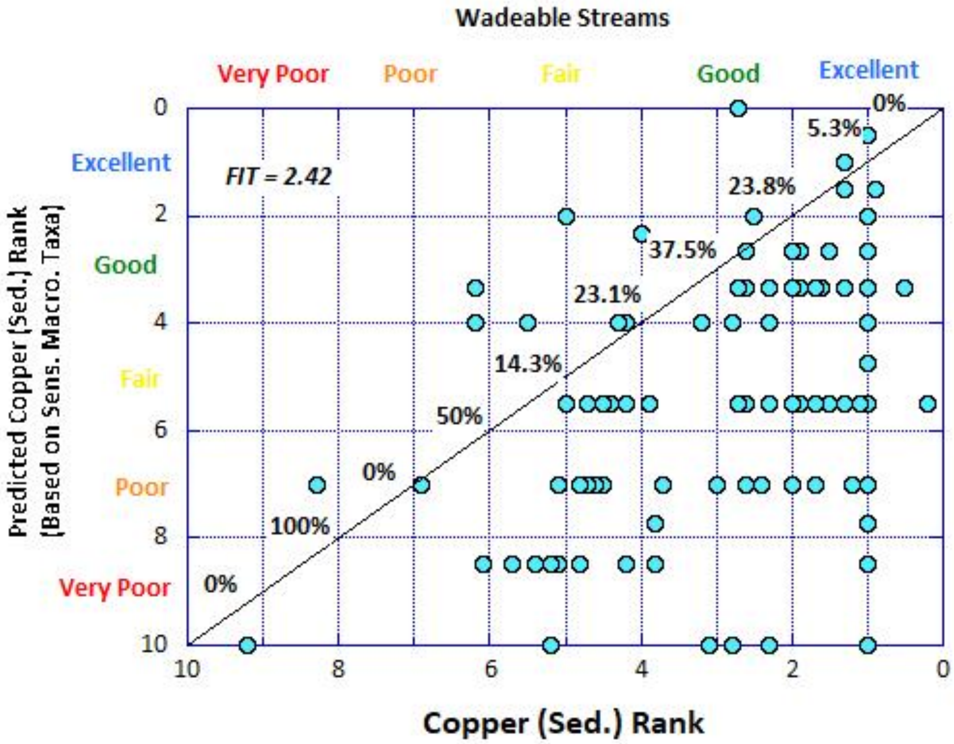



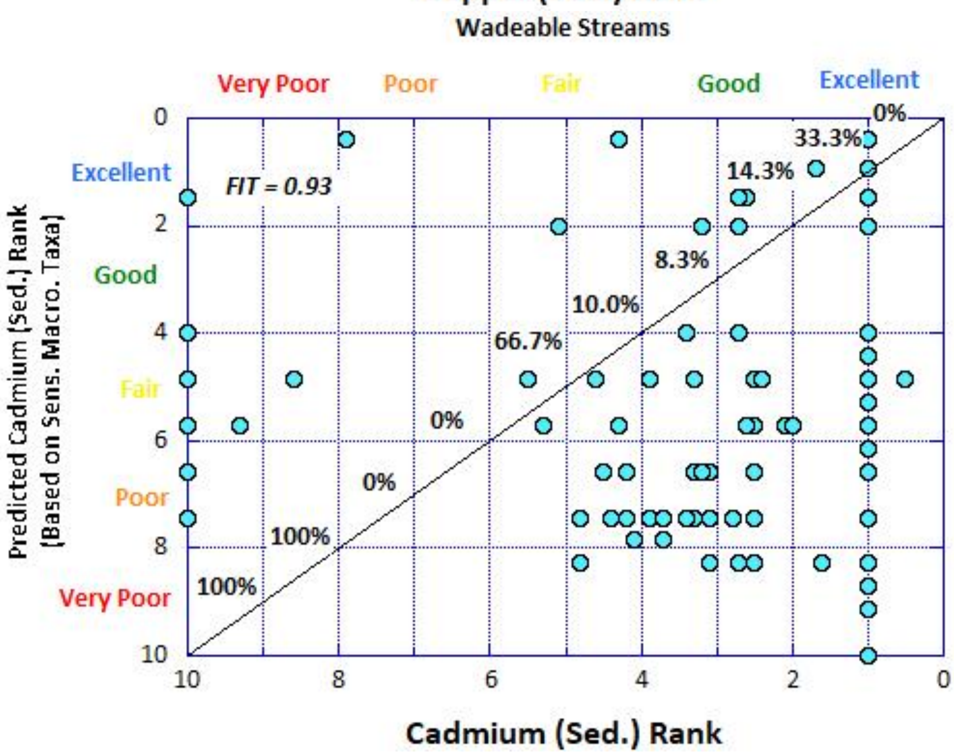

***Appendix Figure C-9**. Plots of Hydro-QHEI rank (top) and Poor Habitat Attribute rank (bottom) vs. predicted stressor ranks based on stressor-specific sensitive fish) for headwater and wadeable stream sites in the IPS study area. Line reflects 1:1 prediction with ambient stressor ranks. Prediction "error" where more sensitive fish species occur than expected based on stressor level for integer levels of stressor effects are presented as percentages. FIT coefficient is a measure that increases with the percent of errors and the magnitude of errors (i.e., magnitude of deviation from expected sensitive fish species).*



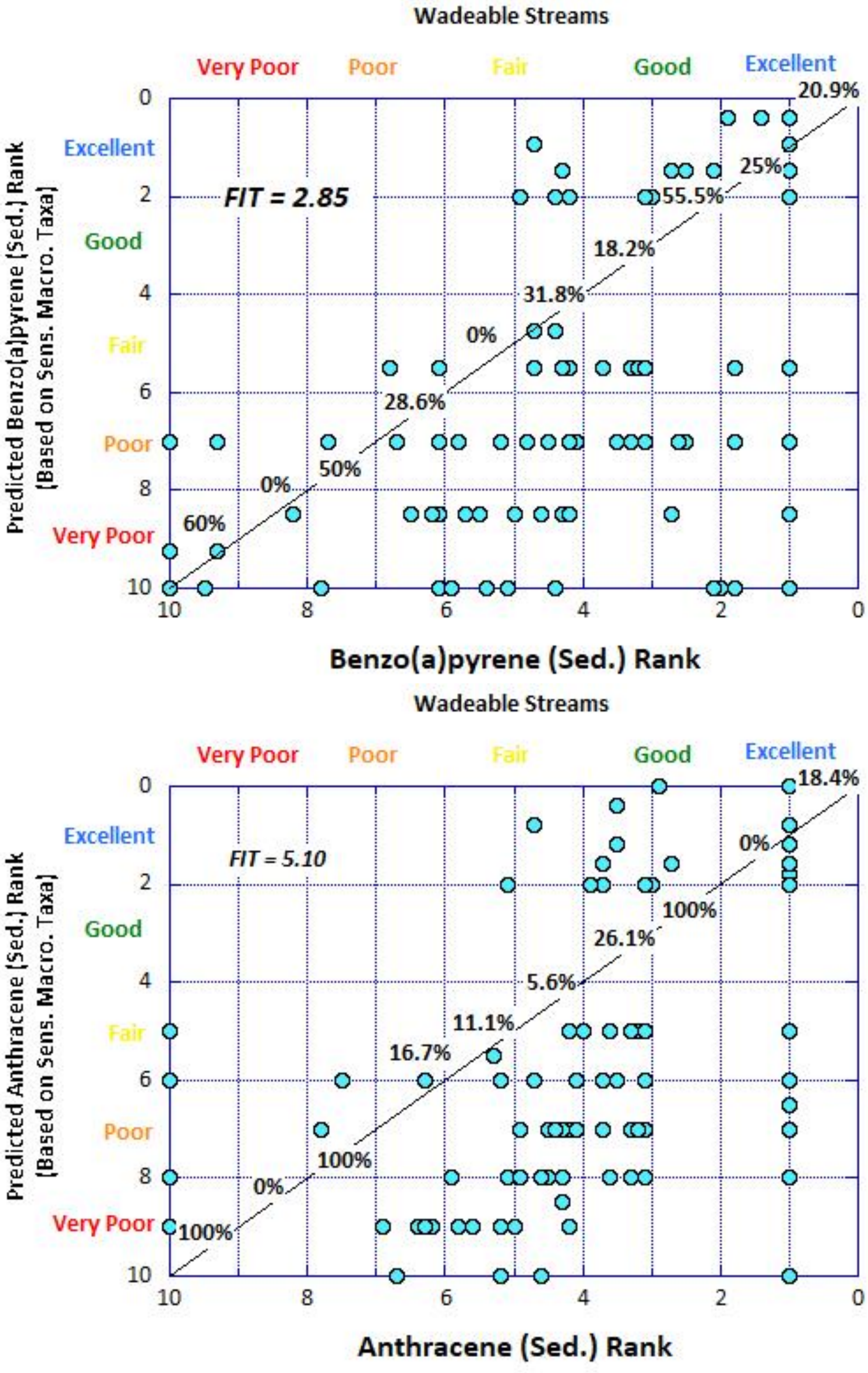




***Appendix Figure C-10****. Plots of lead (water) rank (top) and zinc rank (bottom) vs. predicted stressor ranks based on stressor-specific sensitive fish (zinc) or macros (lead) for headwater and wadeable stream sites in the IPS study area. Line reflects 1:1 prediction with ambient stressor ranks. Prediction "error" where more sensitive fish species occur than expected based on stressor level for integer levels of stressor effects are presented as percentages. FIT coefficient is a measure that increases with the percent of errors and the magnitude of errors (i.e., magnitude of deviation from expected sensitive fish species or macro taxa).*




***Appendix Figure C-11**. Plots of copper (sediment) rank (top) and zinc (sediment) rank (bottom) vs. predicted stressor ranks based on stressor-specific sensitive macros for headwater and wadeable stream sites in the IPS study area. Line reflects 1:1 prediction with ambient stressor ranks. Prediction "error" where more sensitive fish species occur than expected based on stressor level for integer levels of stressor effects are presented as percentages. FIT coefficient is a measure that increases with the percent of errors and the magnitude of errors (i.e., magnitude of deviation from expected sensitive macro taxa).*




***Appendix Figure C-12***. *Plots of Benzo(a)pyrene (sediment) rank (top) and Anthracene (sediment) rank (bottom) vs. predicted stressor ranks based on stressor-specific sensitive macros for headwater and wadeable stream sites in the IPS study area. Line reflects 1:1 prediction with ambient stressor ranks. Prediction "error" where more sensitive fish species occur than expected based on stressor level for integer levels of stressor effects are presented as percentages. FIT coefficient is a measure that increases with the percent of errors and the magnitude of errors (i.e., magnitude of deviation from expected sensitive macro taxa).*